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Background Report IV: Comparison of Risks from Land-Based and Ocean-Based Incineration

Volume II: Appendices

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COMPARISON OF RISKS FROM LAND-BASED AND OCEAN-BASED INCINERATION

VOLUME II: APPENDICES

March 1985

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Appendix A

CALCULATION OF ANNUAL WASTE THROUGHPUT FOR THE VULCANUS II

INTRODUCTION AND SUMMARY

Several factors affect the capacity of ocean incineration systems. In this appendix, we describe these factors and explain the assumptions and calculations used to estimate the annual throughput of the ocean incineration operation proposed by Chemical Waste Management, Inc. (CWM).[1]

Our estimates of annual throughput are functions of two key parameters in the case study:

1. The Vessel: CWM officials have indicated that they intend to use only the I/V Vulcanus II (Vulcanus) for operations in the Gulf of Mexico; her sister ship, M/T Vulcanus, will be called into service only if sufficient (unexpected) demand warrants her operation. The analysis, therefore, is based on the operating capacity of the Vulcanus II.
2. The Wastes: As explained below, the properties of the wastes to be incinerated affect the operation of the incineration system. We have selected two wastes for analysis. The first contains concentrations of polychlorinated biphenyls (PCBs) at the permit specified maximum level of 35%. The primary constituent (50% by weight) of the second waste is ethylene dichloride (EDC). We consider the EDC waste to be representative of many industrial hazardous wastes which would be burned aboard the Vulcanus.

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[1] The analysis of annual throughput for the Vulcanus is based in large part on the vessel operating plan described to us by Bert de Boer of CWM.

This appendix describes how these parameters affect annual throughput by analyzing the following factors:

- the vessel specifications for the Vulcanus;
- the properties of the waste;
- the rated capacity of the onboard incinerators; and
- the operating plan and itinerary of the vessel.

Following this analysis, the appendix summarizes the calculations used to derive annual waste throughput. These calculations yield significantly different annual incineration capacities for different waste materials incinerated aboard the Vulcanus:

- 56,000 MT/yr of wastes containing 35% PCBs
- 68,400 MT/yr of wastes containing 50% EDC.

The appendix concludes with a sensitivity analysis that estimates annual throughput for different waste streams and alternate operating conditions.

SYSTEM PARAMETERS WHICH LIMIT ANNUAL THROUGHPUT

Vessel Specifications

Two aspects of vessel design determine the maximum annual throughput of ocean incineration systems: the volume of the cargo tanks and the maximum net cargo weight of the ship. Annex IV of the "Contingency Plan for the Ocean Incineration Operations of the Vulcanus II" (Chemical Waste Management, Inc. and Ocean Combustion Service, BV., July 1981) reports the volume of the Vulcanus' tanks (3170 cubic meters) and the net cargo capacity of the vessel (approximately 4000 MT). Assuming for safety purposes that the vessel's tanks are never filled beyond 97 percent of capacity, the maximum cargo that the Vulcanus II can carry is either 3075 cubic meters or 4000 MT of waste.2/ The density or specific gravity of the waste material, therefore, determines

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2/The Contingency Plan indicates that the Vulcanus is equipped with alarms that sound when the tanks reach 98 percent capacity. Accoring to Bert DeBoer of CWM, the maximum load carried in practice is 97 percent.

whether the ship's volume or weight capacity is the factor that limits the quantity of cargo carried on each voyage.

Waste Properties

Specific Gravity

As noted above, the specific gravity of a waste material determines whether the vessel operates with full tanks or with partially full tanks but loaded to maximum cargo weight. When the Vulcanus carries liquids with specific gravity of 1.3, both constraints bind; her tanks are full and her cargo weight limit is reached. Thus, liquids with specific gravities less than 1.3 cause the Vulcanus to be operated with full tanks but lighter than maximum cargo; specific gravities greater than 1.3 cause the Vulcanus to operate at her maximum cargo weight but with partially empty tanks.

Heat Content

The heat content of the waste material determines the incineration rate of the material. Incineration rates will be highest for wastes that have heat contents high enough to support combustion without the addition of substantial amounts of auxiliary fuel. Wastes with low heat content must be burned in combination with fuel oil or other materials to ensure that the flame temperature of the incinerators remains greater than 1200 degrees centigrade. Burning non-waste materials to sustain temperature reduces the effective capacity of the incinerators. In contrast, wastes with extremely high heat contents must be burned at reduced rates in order to avoid damage to the refractory walls of the incinerators (the rate at which heat is released by incineration of material must not exceed the heat release capacity of the incinerator). Thus, some wastes which contain significant amounts of oil or other high heat content materials must be burned at reduced rates to avoid damage to the incinerators.3/ The optimum heat content for wastes to be burned aboard the Vulcanus is 5,400 btu/lb.4/

3/Personal communication with D.G. Ackerman, Radian Corporation.

4/"Monitoring of Combustion Efficiency, Destruction Efficiency and Safety During the Test Incineration of PCB Waste, Part I: Combustion and Destruction", TNO, September 15, 1982.

Rated Capacity of Incinerators

The design capacity of each of the three incinerators onboard the Vulcanus is approximately 15 MT/hr of liquid waste (Table 21, "History of Environmental Testing of the Chemical Waste Incinerator Ships M/T Vulcanus and I/V Vulcanus II: Addendum II", TRW Inc, June 1983). In fact, experience in Europe indicates typical incineration rates well below design capacity. CWM officials report that in practice the incinerators on the Vulcanus II each operate at a rate of 5 MT/hr for PCB wastes, and 8-8.5 MT/hr for other chlorinated organic wastes. The lower rate for PCBs, which typically are contained in an oily waste with high heat content, is necessary to avoid damage to the refractory walls. Based on these figures, we employ the following calculations to estimate total hourly waste throughput for the Vulcanus II:

- 3 incinerators * 5 MT/hr/incinerator = 15 Mt/hr of PCB waste;
- 3 incinerators * 8.33 MT/hr/incinerator = 25 MT/hr of chlorinated organic wastes (e.g. EDC).

Vessel Operating Plan/Itinerary

CWM officials related the following operating plan and itinerary for the operation of the Vulcanus in the Gulf of Mexico.

Ship Utilization

CWM plans to operate the Vulcanus 330 days per year; the remaining 35 days will be used for maintenance. Therefore, 7920 hours remain in the year for vessel operation.

Vessel Loading

CWM estimates that loading the Vulcanus will take an average of 180 hours per voyage. This estimate is based on the design of the existing Chickasaw port facility, which lacks storage capacity, and the assumption that CWM will allocate approximately ten tank trucks to transport wastes from its Emelle, Alabama storage facility to the waiting ship. If CWM constructs storage facilities in Chickasaw, loading time could be reduced to 36 hours. Improving the port facility therefore could reduce total voyage time and significantly increase annual waste throughput.

Vessel Transit

Vessel transit between the loading dock and the incineration site consumes hours potentially available for waste incineration. CWM officials report that the Vulcanus would travel at speeds of up to 5 nautical miles per hour (knots) from Chickasaw to Northern Mobile Bay; during the remainder of the trip the vessel would average 12 knots. It is 13 kilometers (km) from the dock to Northern Mobile Bay, and approximately 785 km from the North Bay to the incineration site. From these figures, we estimate an elapsed transit time from the dock to the incineration site of 37 hours (rounding errors in calculation account for discrepancies in Exhibit A-1). The return voyage also takes 37 hours. Thus, the total transit (excluding time spent at the Gulf incineration site) takes 74 hours.

Time in Burn Zone

While at the burn zone, we estimate the Vulcanus spends 90% of the time incinerating waste materials -- the remainder provides allowances for down time. While incinerating wastes, the Vulcanus is confined to the Burn Zone and is required by EPA's draft permit to maintain a minimum speed of 3 knots relative to the wind so that the crew will not be exposed to gases from the incinerator stacks.

Because incinerator feed rates for PCB-containing wastes are lower than for EDC wastes, it takes more time to incinerate a load of PCB-containing waste than it does to incinerate a load of EDC wastes. We estimate the Vulcanus will require 169 hours to incinerate a load of EDC wastes and 296 hours to incinerate a load of PCB wastes.

Summary

The annual throughput of the Vulcanus ocean incineration system is dependent on the time required for ship loading and transit, the vessel's cargo capacity, the properties of the waste cargo, and incinerator capacity and utilization. Considering the factors discussed above, we estimate the Vulcanus will require 423 hours to complete an EDC incineration voyage and 550 hours to complete a PCB voyage; these estimates include loading, transit and incineration times. A discussion of the calculations we performed to derive the estimated annual throughput for 35% PCB waste and 50% EDC wastes follows.

CALCULATION OF INCINERATOR SHIP ANNUAL THROUGHPUT

35% PCB Wastes

The properties which determine the annual throughput for incineration of 35% PCB wastes (see Exhibit A-1: Case A and Assumptions) are as follows:

o	Specific Gravity	= 1.4
o	Cargo Carried	= 4000 MT
o	Loading Time	= 180 hours
o	Transit Time	= 37 hours
o	Incineration Rate	= 15 MT/hr
o	Utilization at Burn Site	= 90%
o	Return Trip	= 37 hours
o	Available Time	= 330 days/year
o	Available Hours in Year	= 7920 hours

To determine the time required for a single voyage, we must calculate the time required to incinerate the cargo and add to that the time spent in transit and in loading the ship. The time required to incinerate the cargo is a function of cargo weight, incineration rate, and utilization at the burn site, and is calculated as follows:

Time to Incinerate PCB cargo [hr] =

weight of cargo (MT) / incineration rate (MT/hr) /

utilization at burn site (%)

We estimate the Vulcanus will require 296 hours to incinerate one load of 35% PCB waste. Therefore, one voyage would require 550 hours (loading time included). We estimate the number of trips that can be completed in a year as follows:

Number of Trips Per Year =

available hours per year / hours per trip =

14.4 trips per year

Rounding down, we estimate the Vulcanus could complete 14 voyages per year, incinerating 4000 MT of PCB waste per voyage. The resulting annual throughput would be 56,000 MT.

50% EDC Wastes

The calculations and assumptions we use in estimating the number of voyages per year for EDC wastes (see Exhibit A-1: Case B and Assumptions) are identical to those used in the PCB waste calculation except for the following changes:

- Specific Gravity = 1.2351
- Cargo Carried = 3170 cubic meters * 0.97 * 1.2351 = 3800 MT
- Incineration Rate = 25 MT/hr

Using these assumptions, we estimate that incineration of a load of EDC waste would require 169 hours; total voyage time would be 423 hours. We project that 18 EDC incineration voyages could be completed in a year; the total annual throughput in an 18 voyage year would be 68,400 MT.

Sensitivity Analysis

Loading Time

Of the factors discussed above, loading time is most likely to be reduced; CWM officials anticipate improvements to the facilities at Chickasaw pending demonstration of sufficient demand for incineration services. Based on CWM statements, the proposed improvements would reduce loading time from 180 hours/voyage to 36 hours/voyage. If all else remained the same the number of incineration voyages for the wastes considered would be:

- o 28 EDC incineration voyages/year (106,400 MT/yr), a 56% increase in annual throughput (Case C)
- o 19 PCB incineration voyages/year (76,000 MT/yr), a 36% increase in annual throughput (Case D).

Waste Density

The specific gravity of the waste material affects the cargo capacity of the vessel, and, therefore, affects the annual throughput of the incineration system. Our calculations above predict the system's capacity for wastes of representative specific gravity. Execution of the throughput calculations for non-PCB wastes with specific gravity greater than 1.3 yields the following estimated throughput:

- o 18 non-PCB incineration voyages/year (72,000 Mt/yr) (see Case E)

Since we assume maximum cargo per voyage (4,000 MT) and maximumm incineration rate (25 MT/yr), this estimate represents the maximum annual throughput for the Vulcanus under current operating conditions.

If we combine improved loading facilities and non PCB wastes with specific gravity greater than 1.3, the annual throughput increases to 108,000 MT/yr in 27 voyages (see Case F). This estimate represents the Vulcanus' maximum annual throughput should CWM upgrade the existing Chickasaw port operation and install waste storage capacity at the dock.

CONCLUSION

Our projections indicate that the annual throughput of CWM's proposed ocean incineration operation depends upon several factors, including the carrying capacity of the Vulcanus, the density and heat content of the waste cargo, the hourly throughput rate of the Vulcanus' incinerators, and the vessel's voyage plan. Our baseline assumptions of system operating parameters yield the following estimates of annual waste throughput:

- o 56,000 MT/yr for PCB wastes;
- o 68,400 MT/yr for EDC wastes.

The throughput of the existing CWM system could increase to 72,000 MT/yr if the Vulcanus' handled relatively dense non-PCB wastes. Annual throughput could increase much more, however, if the Chickasaw port facility were improved. In that event, annual throughput for non-PCB wastes could reach 108,000 MT.

Exhibit A-1

CALCULATION OF VOYAGE PLAN AND ANNUAL THROUGHPUT FOR VULCANUS II

ASSUMPTIONS

NET CARGO CAPACITY:
Metric Tons (Max): 4000
Cubic Meters: 3170
U.S. Gallons: 837500
Utilization of Tank Volume (Max): .97

TOTAL INCINERATOR THROUGHPUT (MT/HR):
PCB Wastes: 15
Non-PCB Chlorinated Organics: 25

VESSEL SPEED (kt):
Loading Dock to North Bay: 5
North Bay to Burn Site: 12

TRIP DISTANCES FROM DOCK (NM):
North Bay: 13
Bay Entrance: 78
Miss. Channel: 258
Burn Site: 800

LOADING TIME (HR):
Existing Facility: 100
Improved Facility: 36

GENERAL OPERATION:
Days/Year: 330
Incinerator Utilization at Burn Site: .9

CALCULATION OF ANNUAL THROUGHPUT

	CASE A		CASE B		CASE C		CASE D		CASE E		CASE F	
	PCB WASTES/EXISTING LOADING (S.G.=1.4) 4000 MT/VOYAGE)		EDC WASTES/EXISTING LOADING (S.G.=1.235) 3800 MT/VOYAGE)		EDC WASTES/IMPROVED LOADING (S.G.=1.235): 3800 MT/VOYAGE		PCB WASTES/IMPROVED LOADING (S.G.=1.4) 4000 MT/VOYAGE)		CHLORINATED WASTES/EXISTING LOAD (S.G.=1.3) 4000 MT/VOYAGE)		CHLORINATED WASTES/IMPROVED LOADING (S.G.=1.3) 4000 MT/VOYAGE)	
ACTIVITY	HOURS	CUMULATIVE HOURS	HOURS	CUMULATIVE HOURS	HOURS	CUMULATIVE HOURS		CUMULATIVE	HOURS	CUMULATIVE HOURS	HOURS	CUMULATIVE HOURS
Loadings	100	100	100	100	36	36	36	36	100	100	36	36
Steam to North Bay:	2	102	2	102	2	38	2	38	2	102	2	38
Steam to Bay Entrance:	3	104	3	104	3	40	3	40	3	104	3	40
Steam to Miss. Channel:	9	193	9	193	9	49	9	49	9	193	9	49
Steam to Burn Site:	24	217	24	217	24	73	24	73	24	217	24	73
Incineration at Burn Site:	296	513	169	386	169	242	296	369	178	395	178	251
Return to Dock:	37	550	37	423	37	279	37	406	37	432	37	288
Total Hours/Voyage:		550		423		279		406		432		288
Available Hours/Year:		7920		7920		7920		7920		7920		7920
Voyages/Year:		14		18		28		19		18		27
Annual Throughput (MT/YR):		56000		68400		106400		76000		72000		108000

Appendix B

TRANSFER AND STORAGE RELEASE ANALYSIS

B.1 INTRODUCTION

This appendix covers the fugitive emissions and episodic spill events associated with the transfer and storage operations for three incineration systems. For spill events, both the frequency of releases and the size of releases are covered. Fugutive emissions are estimated on an annual basis.

The three systems evaluated are:

1. Interim storage at Emelle and transfer to Chickasaw for ship loading;
2. Storage at Chickasaw and transfer to the Vulcanus; and
3. Storage and incineration at a land based facility.

Our analysis estimates the fugitive emissions associated with both storage components and transfer equipment, such as pumps, and the relatively unlikely large scale releases from storage. It does not include releases from transporting wastes, from non-loading incidents involving the incinerator ship, or from the initial loading which occurs at the waste generation sites.

The remainder of this appendix is divided into three sections. Section B.2 discusses the generation of fugitive emissions at each of the facilities, Section B.3 looks at release quantities and frequencies for acute spills at each facility and Section B.4 summarizes the information generated in B.2 and B.3.

B.2 EVALUATION OF FUGITIVE EMISSIONS

B.2.1 Introduction

Fugitive airborne emissions are typically associated with leakage from the seals and fittings of chemical, petrochemical, or petroleum process, transfer, or storage systems. In contrast to the acute health hazards that may be posed by accidental and episodic discharges to the environment, these discharges may potentially pose a long-term chronic health hazard to surrounding populations. For completeness, this discussion encompasses traditional sources of fugitive emissions as well as relatively minor point sources associated with various waste transfer, processing, and storage sytems, all of which are capable of emitting airborne contaminants on a continuous or intermittent basis. It does not. however, address emissions associated with actual incineration of waste materials, this being a major point source considered separately in this report.

The overall analysis addresses several scenarios that vary with respect to the location and timing of waste accumulation, storage, transfer and incineration activities. Since most of the planned or existing facilities associated with these scenarios are common to more than one scenario, the following discussion is on a facility-specific basis. It is followed by a summary of estimated emissions for the various facilities.

The facilities to be considered include:

- The existing truck-to-ship waste transfer facility located in Chickasaw, Alabama;
- The above facility with the addition of two 800,000 gallon waste storage tanks; and
- That portion of the facility in Emelle, Alabama that could or would support the receiving and eventual shipment of wastes to the existing facility in Chickasaw.

The three systems we evaluated are as follows:

- Scenario #1: Wastes are received from generators via tank trucks at Emelle, accumulated in various liquid storage tanks, transferred to tank trucks for transport to Chickasaw when an incinerator ship is in port, and eventually transferred to an incinerator ship docked in Chickasaw. This is essentially the system used in the past to support testing of the incinerator ships.

- Scenario #2: The second scenario envisions the construction of two 800,000 gallon storage tanks at the existing Chickasaw facility. As above, generators would transport wastes to Chickasaw via tank trucks, but the storage tanks would eliminate potential scheduling difficulties and some of the road transportation. Incinerator ships would be quickly loaded from the storage tanks while in port.
- Scenario #3: The final scenario involves a facility resembling that discussed in Scenario #2 in conjunction with a land-based incinerator. (The assumed capacity of storage tanks may be larger than necessary to support a land-based incinerator with a capacity equivalent to the Vulcanus.)

B.2.2 Potential Emission Sources

The potential emission sources considered are:

- Seals and fittings;
- Storage tanks and containers; and
- Transfer facilities.

Seals and Fittings

Sources of fugitive emissions relating to seals and fittings in a waste disposal facility can include (1):

- Drive shaft seals of pumps, agitators, and compressors;
- Stems, bonnets, and flow-seals of in-line or open-ended valves;
- Seats and flange faces of pressure relief valves;
- Face seals of flanges;
- Face or thread seals of measurement devices (pressure, vacuum, temperature, level, etc.); and
- Flanges and valves of sampling assemblies.

Some of these items might directly leak contaminant vapors. Others may discharge liquids from which volatile components may evaporate.

Pumps

According to Rothman (2), the "desired leakage rate" for lubrication through the fibrous compression packings for pumps having "a typical stuffing-box design" is on the order of 8-10 drops per minute. Hoyle (3) explains that a minimum controlled leakage rate of a drop per second for lubrication through the simple packed seal of a 300 gpm pump would equate to 15 cups per day of the material being transferred.

A second type of pump employs a single mechanical seal to reduce the need for lubrication. According to Van Wagenen (1), these pumps have a leakage rate of about one drop per minute and operate effectively for extended periods only if the material being transferred is free of grit or solids.

A third type of pump is fitted with two or more mechanical seals with void space(s) between them. Circulation of a seal fluid through the voids (at a somewhat higher pressure than the product stream) results in substantially less leakage than that associated with single mechanical seals.

Where no leakage can be tolerated, there is also the option of using one of the limited number of canned or diaphragm type sealless pumps available on the market. These have not been adopted widely to date, according to Van Wagenen, probably because of somewhat higher initial cost and lack of long-term performance history.

Valves

In general, valves are placed between lengths of continuous piping or at the end of a length of pipe, thus classifying them as in-line valves or open-ended valves. Potential leakage pathways for all valves include the area around the valve stem and the joining ring to the valve body (the bonnet). Leakage past the closed flow control element within the valve body can be problematic when valves are open-ended.

The first two sources of leakage can be avoided by use of packless type valves that are commercially available, these being of the diaphragm type or the sealed metal bellows type. Leakage from open-ended valves can be controlled by closing the open end with blind flanges, caps, or plugs.

Flanges

Flanges provide the means to join two pieces of pipe or equipment together by inserting a gasket material between two adjacent flange faces. Van Wagenen reports that causes of leakage in flanges include:

- Improper fit resulting from nonaligned flange faces;
- Seal deformation due to stresses from adjacent piping or equipment; and
- Repeated openings without changing the gasket.

One method to greatly reduce or eliminate emissions from flanges involves joining of sections by welding, but this is not always practical. A good maintenance program can also be highly beneficial.

Pressure Relief Valves

Pressure relief valves, as their name implies, provide a means to release excess pressure from a closed chemical processing or storage system. Some such valves are expected to operate only occasionally under extraordinary and/or unusual conditions. Others may operate far more frequently. In either case, an open valve may discharge contaminants directly to the environment. Closed valves may leak due to: 1) imperfect reseating of seal surfaces following a relief cycle; or 2) simmering resulting from pressure fluctuations near the set pressure of the valve.

A method to control normal emissions from these units requires venting of contaminants to a vapor collection and treatment system. Leaks from valves expected to vent infrequently can be prevented by installation of a rupture disk upstream from each valve, these disks being replaced on an "as needed" basis.

Process Sampling Assemblies

The method by which samples are taken from process streams or storage containers can also affect total fugitive emissions. Van Wagenen describes several special devices and procedures to reduce emissions. Others include use of ventilated enclosures at sampling points, with exhaust air directed to a vapor collection and treatment system.

Storage Tanks and Containers

A large modern waste incineration facility may be expected to require numerous bulk storage tanks for different types of liquid wastes, solvents, and fuels. The major types worthy of consideration include pressure tanks, fixed-roof tanks, floating-roof tanks, and conservation tanks.

Pressure tanks are designed to operate at pressures above ambient conditions, and may be considered gastight (except when pressure relief valves are activated). It is noted, however, that few industrial wastes may require storage in vessels of this type based solely on vapor pressure considerations.

Danielson (4) reports that the term gastight is often applied to ordinary, vertical, cylindrical, welded fixed-roof tanks, but that the term is misleading. Many such tanks have free vents open to the atmosphere while others have vents that open at slight positive pressures. Such tanks have many standard fittings including gaging hatches and sampling hatches that may also become sources of leakage.

Floating-roof storage tanks are most commonly used for volatile materials with vapor pressures in the lower explosive range. Several variations have the common feature of a roof that floats up and down with internal liquid level.

Conservation tanks include lifter-roof tanks and fixed-roof tanks with internal flexible diaphragms or internal plastic floating blankets. Lifter-roof tanks may be used for storage of low-pressure gaseous products or for storage of liquids with low vapor pressures. They may also be used as vapor surge tanks when manifolded to the vapor spaces of fixed-roof tanks.

Major causes of emissions from fixed- and floating-roof tanks include:

- Changes in temperature and weather conditions;
- Filling operations; and
- Wicking and wetting effects.

As the environment warms during daylight hours and cools at night, the vapor space within fixed roof tanks changes pressure with respect to external conditions, typically resulting in outflow of emissions during the day and inflow of fresh air at night. This diurnal variation is referred to as the tank's breathing cycle.

Wind blowing past a free-vented tank may entrain or educt vapors from the tank or its fittings

As a fixed-roof tank is filled with liquid, the vapor space within the tank decreases in volume and excess internal air volumes vent to the outdoors. Extraction of liquid from the tank results in entrance of fresh air which in turn promotes evaporation of tank contents.

Wicking and wetting emissions are primarily associated with floating-roof tanks. Wicking emissions result from capillary flow of liquid between the outer side of the sealing ring on the roof and the inner wall of the tank. Wetting emissions are attributable to evaporation from wet walls exposed to the atmosphere during extraction of tank contents.

A great many factors influence the specific rate of emissions from various type of tanks. Estimation procedures for specific applications may be found in References 4 and 5. Potential control measures, also discussed in these references, include edge seals for floating-roof tanks, use of floating plastic blankets or microspheres above liquid surfaces, installation of vapor balance systems, installation of vapor recovery and treatment systems, and a variety of miscellaneous measures.

Transfer Facilities

Any waste disposal facility for bulk liquids will require one or more systems for receiving and unloading tank vehicles (highway or rail), tank barges, and/or trucks or other vehicles transporting drummed or otherwise containerized wastes. Each of these systems is potentially associated with various sources of fugitive emissions as connections are made and broken, as valves are opened and closed, and as non-gastight containers await emptying. Potential control measures vary with the specific details of the operation, but include dry disconnect couplers on transfer arms or hoses, the use of blind flanges on lines when and where necessary, and the use of vapor return and/or recovery systems.

Sites serving incinerator ships additionally require a marine transfer facility. Emissions from transfer operations can potentially be controlled via use of articulated loading arms with vapor shields and closed-loop vapor return lines, as described by McNeil et al. (6).

B.2.3 Present Facility - Chickasaw, Alabama

The present facility of Chemical Waste Management Inc. in Alabama is designed to receive liquid waste materials via tank trucks for direct loading into incinerator ships. The following describes the various subsections of the overall system with special reference to potential emission sources. Also considered are waste characteristics and the probable magnitude of overall emissions. Information for all but the latter topic was derived from References 7-9.

Tank Vehicle Unloading Facility

Tank trucks of 5000 gallon capacity are received at Chickasaw within a four-bay transfer facility capable of simultaneously unloading four vehicles at pumping rates from 200 to 800 gpm. Transfer is accomplished via four OPW G32F Bottom Unloading Arms that are attached with swivel joints to a common eight-inch pipeline suction header. Each arm can be isolated by a four-inch butterfly valve. Four-inch dry disconnect couplers with Viton seals on the ends of the arms serve to eliminate drippage and vapor emissions when arms are disconnected from trucks. All valves here and elsewhere have teflon seals that are claimed to prevent leakage.

The only apparent and potentially significant source of fugitive emissions to be considered involves the necessity for operating personnel to gage each truck manually to determine when it is empty. This occurs once or twice during the unloading of each truck and involves opening of a manhole cover and use of a dipstick. The cover is opened for one or two minutes at a time. After being read, the wet dipstick is wiped down with a rag that is subsequently stored in a drum for eventual disposal.

It is noteworthy that components of the transfer system, including those that follow, are designed for a pressure capacity of 150 psig. Normal line pressures will be below 50 psig. The entire system will receive frequent inspections to detect leakage.

Pumping Facility

The above transfer facility is directly adjacent to a pump building housing two 400 gpm positive displacement transfer pumps equipped with explosion-proof motors and mechanical seals. The two pumps serve a common eight-inch flanged and welded header.

- Each pump can be isolated using upstream and downstream valves.
- The suction side of each pump is equipped with a strainer having a valved vent and drain.
- Recycle lines and pressure switches ensure that line pressures are maintained within desired limits.
- The pump seals do not permit visible leakage.

Pipeline to Dock

Transfer piping between the pumping facility and the dock consists of eight- and six-inch diameter sections of welded steel pipe. As noted previously, this type of system has the lowest probability of developing leaks.

Dock to Ship

At the dock, a continuous 50-ft length of six-inch flexible hose, suspended between the dock and the ship and supported by a hose bridle, connects the rigid pipe to the ship. This hose is tested to 200 psi on an annual basis. Isolation valves are provided on the loading platform and at the hose connection to the ship, which is blinded off until connected to the ship and re-blinded upon disconnection. Removal and reinstallation of the blind flange is accomplished over a drip pan, but any fluid in the line at this point (at the completion of loading) would only consist of a clear oil used for decontamination purposes (see below).

In order to minimize fugitive emissions from cargo tanks during loading operations, and thereafter, shipboard tanks are vented through an onshore 55-gallon drum filter filled with activated carbon, this being replaced every 7-10 days to ensure maintenance of a high filtration efficiency. All manhole covers on the tanks are kept closed and are tightly fitted with teflon seals.

Termination of Loading

Upon completion of ship loading, the transfer system will be decontaminated by pumping approximately 1250 gallons of clear diesel oil or a similar material from each station of the truck receiving facility through the system and into the ship. Residual material will then be drained from the system via a vacuum truck from two low points. One of these is at a sump in the receiving facility. The other is located at a pipe bridge over an access roadway. The overall procedure will be repeated until the residual PCB content of the waste (if any) is below 50 ppm. All drainage will be returned by truck to the CWM facility in Emelle, Alabama.

Waste Characteristics

The mixed organohalogen wastes expected to be authorized for incineration at sea will be complex mixtures of numerous compounds. While it is not possible to list every waste that may be incinerated, it is feasible to describe certain characteristic wastes that have been successfully incinerated in the past.

Table 1 presents the composition of EDC tars originating as heavy ends from the integrated production of ethylene dichloride and vinyl chloride. Table 2 describes the light wastes from the above process, while Table 3 addresses waste resulting from production of vinyl chloride. Table 4 lists the composition of a collected and blended waste. Operating personnel have further indicated that some wastes may contain up to 30 percent PCBs.

Typical Emission Factors

Of the many types of facilities studied in the past for the purpose of defining emission factors, petroleum refineries most closely resemble the Chickasaw site in terms of equipment and materials handled. Table 5, extracted verbatim from Reference 1, presents estimated emission

TABLE 1

COMPOSITION OF CHLORINATED HYDROCARBON BYPRODUCTS FROM EDC TARS

Description

A black pumpable organic liquid saturated with water, originating as heavy ends from the integrated process of ethylene dicloride and vinyl chloride manufacture.

Composition	% w/w (approximate)
1,2 Dichloroethane	1.5
1,1,2 Trichloroethane	46.0
Dichlorobutene	0.7
1,1,1,2 Tetrachloroethane	8.8
Dichlorobutene isomer	0.6
1,1,2,2 Tetrachloroethane	15.7
Pentachloroethane	6.3
Dichlorodiethyl ether	1.0
Hexachloroethane	10.3
1,3,5 Trichlorobenzene	0.5
1,2,4 Trichlorobenzene	1.9
1,2,3,5 Tetrachlorobenzene	0.2
1,2,3,4 Tetrachlorobenzene	0.3
Pentachlorobenzene	1.2
Others	5.0
Total	100.0

TABLE 2

EDC LIGHTS

Description

Chlorinated hydrocarbon solvent containing partially polymerised chloroprene. An amber colored viscous liquid saturated with water.

Composition	% w/w (approximate)
Carbon Tetrachloride	25.2
Chloroform	25.0
1,2 Dichloroethane	24.6
1,1 Dichloroethane	6.5
Trans 1,2 Dichlorethylene	2.5
Trichloroethylene	2.1
Cis 1,2 Dichloroethylene	1.8
1,1 Dichloroethylene	1.5
Chlorprene isomer	1.4
Vinyl Chloride	0.9
Ethyle Chloride	0.4
Total	91.9

TABLE 3

COMPOSITION OF WASTE CHLORINATED HYDROCARBON

Description

Pumpable organic liquid known as mixture of light and heavy ends from vinyl chloride manufacturing process.

Composition

1,2 Dichloroethane	ca	65%
1,1 Dichloroethane		14%
Dichlorethylenes		8%
Trichloroethylene		4%
Trichloroethylene		6%
Others		3%
Total		100%

TABLE 4

Description

Weight content of the main organic components of the lower phase of the waste liquid (chlorinated hydrocarbons part) for a cargo that has been incinerated. Percentages refer to a water-free solution.

Chemical Name	% weight
Acetone	1
Methylene Chloride	2
2 Butanone	2
Chloroform	8
Ethyl acetate	2
Tetrahydrofuran	2
1,2 Dichloroethane	2
1,1,1 Tribromoethane	2
Benzene	3
1,2 Dichloropropane	2
Ethylene trichloride	4
1,1,2 Trichloroethane	10
Toluene	15
1,1,2,2 Tetrachloro ethylene	7
Butylacetate	3
Chlorobenzene	3
Ethyl Benzene	2
Xylene	5
Xylene + 1,1,2,2 tetrachloroethane	2
Chloro methyl benzene	5
Chloro methyl phenol	5
Dichloro methyl benzene	2
Tetrachlorobenzene	7
Trichlorobenzene	1
Others	2
Total	99

TABLE 5

SUMMARY STATISTICS AND ESTIMATED VAPOR EMISSION FACTORS FOR NONMETHANE HYDROCARBONS FROM BAGGABLE SOURCES (PETROLEUM REFINERIES)

SOURCE TYPE	TOTAL SCREENED†	NUMBER LEAKING§	PERCENT LEAKING	95% CONFIDENCE INTERVAL FOR PERCENT LEAKING	PERCENTILE FOR INDIVIDUAL LEAKS (lb/hr) 1st	99th	EMISSION FACTOR ESTIMATE (lb/hr-source)	95% CONFIDENCE INTERVAL FOR EMISSION FACTOR (lb/hr-source)	EMISSION FACTOR PERCENTILE
Valves									
Gas/Vapor Streams	683	200	29.3	(25.9, 32.7)	0	0.61	0.047	(0.027 , 0.084)	94.4
Light Liquid/Two-Phase Streams	1019	372	36.5	(33.6, 39.5)	0	0.31	0.023	(0.016 , 0.034)	89.7
Heavy Liquid Streams	522	35	6.7	(4.6, 8.9)	0	0.08	0.0007	(0.0002, 0.002)	95.8
Pump Seals									
Light Liquid Streams	470	300	63.8	(58.9, 68.8)	0	4.0	0.26	(0.17 , 0.39)	89.8
Heavy Liquid Streams	292	66	22.6	(17.1, 28.1)	0	0.87	0.045	(0.02 , 0.11)	89.4
Flanges (All)	2030	62	3.1	(2.2, 3.9)	0	0.008	0.00058	(0.0002, 0.001)	97.4
Gas/Vapor	369	10	2.7	(0.8, 4.6)	0	0.008	0.0005	(10^{-5} , 0.005)	
Light Liquid/Two-Phase Streams	616	33	5.4	(3.3, 7.4)	0	0.011	0.0005	(0.0002, 0.001)	
Heavy Liquid Streams	325	6	1.9	(0.2, 3.5)	0	0.009	0.0007	(10^{-5} , 0.02)	
Compressor Seals									
Hydrocarbon Service	145	102	70.3	(62.9, 77.7)	0	17.0	0.98	(0.46 , 2.0)	83.4
Hydrogen Service	85	69	81.2	(71.2, 88.8)	0	2.0	0.10	(0.04 , 0.24)	86.5
Drains (All)	255	49	19.2	(14.4, 24.0)	0	1.3	0.070	(0.02 , 0.20)	93.3
Light Liquid/Two-Phase Streams	100	26	26.0	(14.6, 37.4)	0	1.9	0.085	(0.02 , 0.32)	
Heavy Liquid Streams	107	19	17.8	(9.5, 26.1)	0	0.63	0.029	(0.003 , 0.21)	
Relief Valves (All)	148	58	39.2	(25.1, 40.6)	0	4.9	0.19	(0.06 , 0.52)	91.9
Gas/Vapor Streams	92	42	45.6	(35.2, 56.4)	0	4.0	0.36	(0.10 , 1.30)	
Light Liquid/Two-Phase Streams	28	7	25.0	(10.7, 44.9)	0	0.38	0.013	(0.001 , 0.23)	
Heavy Liquid Streams	23	8	34.8	(16.4, 57.3)	0	0.42	0.019	(0.001 , 0.20)	

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*Some streams could not be accurately classified into stream category, so totals are not the sum of the stream groups.

§Leaking sources in this report are defined as sources with screening values greater than or equal to 200 ppmv or sources with measured leaks greater than 0.00001 lb/hr.

factors for a variety of equipment and materials found at refineries. The material types are defined as follows:

- Gas/vapor streams - materials completely vaporized at process conditions.
- Light liquid/2-phase streams - light hydrocarbon liquids.
- Heavy liquids - primarily kerosene and heavier hydrocarbon liquids.

In terms of the volatility of materials handled, the characteristic waste materials described in Tables 1-4 have several components that are generally more volatile than kerosene but somewhat less volatile than light hydrocarbon liquids. It follows that emission factors for any leaking and otherwise uncontrolled components are likely to lie between those shown for light and heavy liquids.

Table 6, also extracted from Reference 1, presents the distribution of measured leak rates from refinery equipment. Table 7 lists factors for controlled and uncontrolled emissions as listed in Reference 5.

Expected Emissions at Chickasaw

For a variety of reasons, it is impossible to develop a highly accurate estimate of fugitive and related contaminant emissions from present and future operations conducted at the Chickasaw site in the absence of actual measurements. It is, however, feasible to discuss the likelihood of these emissions being significant and to make a first approximation of their order of magnitude. It is stressed that the effort requires several major assumptions and that results are subject to considerable uncertainty.

At the truck receiving facility, the most likely source of emissions involves the manual tank gaging operation utilizing dipsticks. Given that about 151 or 163 tank trucks must be unloaded for each voyage of an incinerator ship under the options being considered, and that there would be respectively 14 or 18 ocean voyages per year using this cargo transfer method, the gaging operation might be conducted a total

TABLE 6

DISTRIBUTION OF MEASURED LEAK RATES †‡
(PETROLEUM REFINERIES)

	VALVES						PUMP SEALS			
Leak Range (lb/hr)	Gas/Vapor Streams		Light Liquid/ Two-Phase Streams		Heavy Liquid Streams		Light Liquid Streams		Heavy Liquid Streams	
	A	B	A	B	A	B	A	B	A	B
>1.0	1.0	68.4	0.0	13.8	0.0	0.0	4.0	70.5	0.0	0.0
0.1-1.0	2.6	22.7	3.3	58.8	0.2	33.6	15.7	24.8	5.5	73.2
0.01-0.1	8.6	7.9	11.5	23.7	0.9	48.0	22.5	4.3	9.6	25.6
0.0001-0.01	8.9	0.9	13.4	3.5	2.7	16.9	16.4	0.4	5.8	1.2
0.00001-0.001	8.1	0.1	8.1	0.2	2.9	1.5	4.9	0.0	1.7	0.0
>0.00001										
	29.2%	100%	36.3%	100%	6.7%	100%	63.5%	100%	22.6%	100%

	COMPRESSOR SEALS				FLANGES		DRAINS		RELIEF VALVES	
Leak Range (lb/hr)	Hydrocarbon Service		Hydrogen Service		All Stream Groups		All Stream Groups		All Stream Groups	
	A	B	A	B	A	B	A	B	A	B
>1.0	15.9	74.2	0.0	0.0	0.0	0.0	1.5	61.6	3.4	76.0
0.1-1.0	33.1	24.3	16.5	75.6	0.2	63.2	4.7	33.0	10.1	19.2
0.01-0.1	16.6	1.4	25.9	22.5	0.6	30.1	6.7	4.9	14.9	4.5
0.001-0.01	4.8	0.1	24.7	1.8	1.4	6.0	5.1	0.5	7.4	0.3
0.00001-0.001	2.1	0.0	14.1	0.1	0.9	0.7	1.2	0.0	2.7	0.0
>0,00001										
	72.5%	100%	81.2%	100%	3.1%	100%	19.2%	100%	38.5%	100%

†Most sources were bagged and sampled to obtain leak rates; some were estimated using procedures described in Section 1 of Appendix A, Reference 16.

‡Column A = Percent of total sources screened with sampled leak rates within leak range.
Column B = Percent of total mass emissions attributable to sources within leak range.

TABLE 7

FUGITIVE HYDROCARBON EMISSION FACTORS FOR PETROLEUM REFINERIES
EMISSION FACTOR RATING: D

Emission source	Emission factor units	Uncontrolled emissions	Controlled emissions	Applicable control technology
Process drains and waste water separators	lb/10^3 gal. wastewater	5	0.2	Vapor recovery systems and/or separator covers
	kg/10^3 liters wastewater	0.6	0.024	
	lb/10^3 bbl refinery feed [c]	200	10	
	kg/10^3 liters refinery feed	0.6	0.03	
Cooling towers	lb/10^6 gal. cooling water	6	NA [d]	Minimization of oil leaks into cooling water system through good housekeeping and maintenance
	kg/10^6 liters cooling water	0.7	NA	
	lb/10^3 bbl refinery feed	10	NA	
	kg/10^3 liters refinery feed	0.03	NA	
Pipeline valves and flanges	lb/day-valve	0.15	NA	Good housekeeping and maintenance
	kg/day-valve	0.07	NA	
	lb/10^3 bbl refinery feed	28	NA	
	kg/10^3 liters refinery feed	0.08	NA	
Vessel relief valves	lb/day-valve	2.4	Neg	Rupture discs up stream of relief valves and/or vent to blowdown system
	kg/day-valve	1.1	Neg	
	lb/10^3 bbl refinery feed	11	Neg	
	kg/10^3 liters refinery feed	0.03	Neg	
Pump seals	lb/day-seal	5	3	Mechanical seals, dual seals, purged seals
	kg/day-seal	2.3	1.4	
	lb/10^3 bbl refinery feed	17	10	
	kg/10^3 liters refinery feed	0.05	0.03	
Compressor seals	lb/day-seal	9	NA	Mechanical seals, dual seals, purged seals
	kg/day-seal	4	NA	
Asphalt blowing	lb/10^3 bbl refinery feed	5	NA	Scrubber, incinerator
	kg/10^3 liters refinery feed	0.014	NA	
Blind changing	lb/ton of asphalt	60	Neg	Line flushing, use of "line" blinds, blind insulation with gate valves
	kg/metric ton of asphalt	30	Neg	
	lb/10^3 bbl refinery feed	0.3	Neg	
	kg/10^3 liters refinery feed	0.001	Neg	
Miscellaneous: sampling, non-asphalt blowing, (sweetening), purging, etc.	lb/10^3 bbl refinery feed	10	NA	Good housekeeping and maintenance
	kg/10^3 liters refinery feed	0.03	NA	
Storage	See Section 4.3			
Loading	See Section 4.4			

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[a] Overall, less than 1 percent by weight of total hydrocarbon emissions are methane

[c] Refinery feed is defined as the crude oil feed rate to the atmospheric distillation column

[d] NA - These factors are not available

of 3200 times per year in the first case and 4400 times in the second (assuming 1.5 gagings per truck on average). Given the further assumption that each event entails loss of approximately 10 grams of volatile material, emissions would respectively total 32 kg per year or 44 kg per year.

The data of Table 5 indicate that approximately 22.6 percent of refinery pumps experience leakage when handling relatively heavy liquids. Given that only two pumps are used at Chickasaw, that they have mechanical seals, that they will receive frequent inspections, and that visual leakage has not been observed to date, it is concluded that significant emissions are unlikely. Nevertheless, with the assumption that one pump might experience some degree of leakage for a short period of time each year due to seal malfunction (say two weeks per year at 0.1 lb per hour), it is possible to estimate annual leakage of approximately 15 kg.

There are on the order of a dozen valves in the transfer system at Chickasaw. The data of Table 5 suggest that one of every 16 valves in refineries on average have measureable leaks when handling heavy liquids. An assumption that one of the valves at Chickasaw will leak at a rate of 0.01 lb per hr provides a total leakage estimate of 0.82 kg over 180 hours of expected transfer operations associated with each voyage. The totals for 14 and 18 voyages are then respectively 11 kg and 15 kg.

The flanges of pipe connections at the Chickasaw facility are welded. It follows that leakage should be negligible. There are no drains, relief valves, or compressor seals of concern.

The liquid level within individual cargo tanks can be determined by a closed, radar-based gaging system. This and the fact that the cargo is not sampled again aboard the ship suggest that the crew has no reason under normal operating conditions to seek direct access to a cargo tank while the ship is in port.

During loading of the ship, a vapor return system directs displaced vapors from individual tanks to a single 55-gallon drum filter containing activated carbon. When in use, this filter is replaced every 7 to 10 days. However, no other specific steps are taken to ensure that filters do not experience "breakthrough" while in service. (Note: A carbon filter permits "breakthrough", i.e., unrestricted passage of contaminant vapors, when the finite capacity of the carbon to adsorb contaminants is exhausted. The specific efficiency and capacity of a carbon filter is a function of the weight of available carbon, the specific nature and concentration of contaminants in the air being filtered, and environmental conditions.)

Given that each ship will be loaded with approximately 800,000 gallons of liquid waste, and assuming:

- The air in cargo tanks during the loading process contains contaminant vapors at an average concentration of five percent of their saturated concentration;
- The carbon filter is 90 percent efficient on average for capture of contaminants;
- The carbon filter is changed sufficiently often to prevent "breakthrough";
- Contaminant vapors have a molecular weight approximately four times that of air; and
- Contaminants have an average vapor pressure of 80 mm Hg at ambient storage temperatures.

It is possible to estimate that 7.5 kg of gaseous emissions might be associated with each loading of an incinerator ship. It is stressed that this result is a first approximation and subject to considerable uncertainty. The actual amount could be significantly greater or less depending upon the efficiency of the carbon filter and the actual average degree of saturation in the vapor space of the cargo tanks (which are continuously inerted by an automatic shipboard inert gas generation plant). The total emissions for 14 voyages are therefore estimated to be 105 kg; for 18 voyages, the total is 135 kg.

B.2.4 Modified Facility - Chickasaw, Alabama

Personnel of Ocean Combustion Services BV have indicated (7) the ultimate desirability of installing two floating roof storage tanks of 800,000 gallon liquid waste capacity at Chickasaw. The total capacity provided would be equal to two full loads for an incinerator ship and would permit loading of a ship from one full onshore storage tank while the second was receiving wastes for a subsequent voyage. Thus, the facility could continually receive waste liquids without having to schedule deliveries to coincide with times when a ship is in port.

Tank Design

For the purposes of this analysis, it is assumed that the Chickasaw facility would be provided with covered-type floating roof tanks. These are essentially fixed-roof tanks with an internal pan-type roof that floats upon the liquid contents. The outer fixed roof protects the internal floating roof from the weather such that no provisions are necessary for rain or snow removal or for protection of the sliding seals that are common between the edge of the internal roof and the tank walls. Vertical guide rods typically prevent rotation of the internal roof and thereby maintain roof alignment. The space between the fixed and floating roof is commonly vented to the atmosphere to prevent formation of a flammable mixture in hydrocarbon service, but Ocean Combustion Service personnel (7) have indicated that this will not be the case at Chickasaw. Although details are not available, it is expected that an appropriate emission control system would be installed to minimize or eliminate contaminant emissions. It is therefore assumed that such a control system would utilize activated carbon filters of 90 percent average efficiency. Further, it is assumed that the tanks would otherwise evolve emissions at rates similar to those of tanks without emission control systems.

Tanks of the overall design of concern reduce evaporative storage losses vis-a-vis more typical fixed roof tanks by minimizing vapor spaces. The American Petroleum Institute (API) has designated the term "covered floating" roof to describe such tanks when the internal roof is of the steel pan-type. The term "internal floating cover" is used for internal roofs constructed of materials other than steel (5).

Tank Emissions

According to EPA's compilation of air pollutant emissions factors (5), there are no correlations available that specifically pertain to tanks with internal floating covers or covered floating roofs. In the absence of such correlations, the API has recommended use of relationships developed for standing storage losses and withdrawal losses from open floating roof tanks. Additionally, it has specified that the user should assume a wind speed of 4 mph when applying these relationships to the subject tanks.

Application of the relationship for standing storage losses requires several assumptions or observations, these being:

- Molecular weight of vapor in tank = 4 x 29 = 116
- Vapor pressure = 80 mm Hg = 1.55 psia
- Tank diameter = 60 ft = 18.29 m
- Average wind velocity = 4 mph = 1.79 m/s
- Tanks are welded with pan or pontoon roof, single or double seal. (Note: Other designs have higher standing storage emissions by a factor of 2.4 to 3.1.)
- Seals are of tight fitting modern design, and
- Tank is gray or aluminum in color. (Note: A color of white would reduce these emissions by 10%.)

Given the above data, it is estimated that each 800,000 gallon storage tank would evolve 6.0 kg/day of gaseous contaminant emissions on average due to standing storage losses. The 90 percent carbon filter efficiency assumed above suggests that no more than 0.6 kg/day would enter the atmosphere.

The relationship available for estimating withdrawal losses was developed specifically for gasoline and is of uncertain applicability for other liquids. Assuming that the tank is constructed of steel (as opposed to gunite-lined, which increases emissions by a factor of 50) and that the liquid waste has a specific gravity of 1.4, provides an emissions estimate of 31.7 kg for each time a tank is completely filled and emptied. However, given the substantially greater overall volatility of gasoline components than components of expected liquid wastes, it is considered reasonable to divide the above result by a factor of about two (giving approximately 16 kg per loading/unloading cycle) to obtain a more representative result. Application of a 90 percent filter efficiency means that about 1.6 kg would enter the atmosphere during each cycle. A total of 14 cycles supporting 14 ocean voyages would therefore entail total emissions on the order of 22 kg. The estimated total for 18 voyages would be 29 kg.

The overall total of standing storage and withdrawal losses from the two tanks is estimated to be 460 kg per year in support of 14 ocean voyages. For 18 voyages, the total becomes 467 kg.

Other Emissions

The availability of storage tanks at Chickasaw would mean that a ship could be loaded considerably more quickly than in the alternative case previously considered. Indeed, it is estimated that loading would require only 36 hours versus the 180 hours associated with receiving waste liquids directly from tank trucks. The shorter time interval would impact on certain prior estimates, as would the presence of ancillary equipment associated with the storage tanks.

Assuming that approximately four additional pumps would be installed at Chickasaw, to permit intertank transfers and transfers between the storage tanks and the ship, permits estimate of an additional 15 kg of emissions per year associated with leaking pump seals. This assumes that one of the four pumps will develop a slow unobserved leak over a period of two weeks each year at a rate of 0.1 lb per hour.

The presence of the two tanks would also suggest the need for additional piping and valves. This equipment is likely to be in a continual service; pipe connections will be welded. If it is assumed that only one of the additional valves leaks at the very slow rate of 0.01 kg per hour, yearly emissions of approximately 40 kg can be estimated.

The pipes and valves between the storage tanks and the ships will be in service for a fewer number of hours each year under this scenario than was the case for piping and valves between the truck transfer facility at Chickasaw and the ship. With 14 voyages per year, a ship loading time of 36 hours, and an assumed valve leakage rate of 0.01 lb/hr, emissions from one leaking valve are estimated to be 2.3 kg per year. For 18 ocean voyages, the yearly total becomes 2.9 kg. These figures are to be compared to the 11 and 15 kg estimates presented for the case of the existing Chickasaw facility.

B.2.5 Existing Facility - Emelle, Alabama

The Chemical Waste Management (CWM) facility at Emelle covers 80 to 90 acres and supports a variety of waste recycling and disposal activities that are unrelated to potential onshore or offshore incineration operations. Consequently, CWM personnel (10) were interviewed to determine how the company would utilize its Emelle facility to support loading of an incinerator ship at Chickasaw. These contacts revealed that the Emelle site is in the process of considerable modification at present and that the topic can only be addressed in general terms.

Tank Truck Unloading Operations

It is assumed that the liquid wastes are transported to Emelle solely by tank truck and that the unloading activity entails emissions similar to those estimated for the analogous activity at Chickasaw. For

14 ocean voyages per year, these were estimated to involve a total of 32 kg of emissions due to tank gaging operations and 15 kg due to pump seal leaks. The figures for 18 voyages per year were respectively 44 kg and 15 kg.

Transfer to and from Storage Tanks

All metal piping systems at Emelle have welded connections and are double piped to contain any possible leakage. Certain other systems are constructed of high-density polyethylene with connections that are fusion welded. Such systems are unlikely to develop significant leaks.

The various piping, pumps, and storage tanks will be associated with numerous valves. Using the data presented earlier for refinery operations, it is assumed that all valves are in continuous service and that approximately two will experience leakage at a rate of 0.01 lb per hour. Over the course of 365 days, total emissions are therefore estimated to be on the order of 80 kg from this source.

Storage Tank Emissions

CWM personnel (10) have indicated a strong preference for a number of smaller tanks at Emelle than the two large tanks desired for installation at Chickasaw. It was mutually agreed that four tanks of approximately 200,000 gallon capacity would be fairly adequate and reasonable for support of incinerator ship operations, thus permitting some degree of blending and mixing of wastes. Given the length of time a ship would be in port under this scenario, and the time required to transfer wastes from Emelle to Chickasaw via tank truck, it would be rare for all of these tanks to be filled when a ship is not in port and additional wastes are delivered to Emelle.

The Emelle facility has traditionally utilized fixed-roof tanks with closed gaging systems. Manholes are not normally opened. Tank emissions are vented through activated carbon filters that are tested periodically and replaced if found to have an efficiency of 90 percent or less. It is assumed for the purposes of this analysis that similar procedures and equipment would be used for storage of wastes intended for incineration.

CWM personnel have noted that their largest tanks have a height of approximately 30 feet. It is therefore estimated that each 200,000 gallon capacity storage tank would have a diameter of approximately 35 feet.

The EPA (5) reports that fixed-roof storage tanks potentially have emissions associated with:

- Breathing losses;
- Working losses associated with tank filling; and
- Working looses associated with emptying of tank contents.

Estimation of breathing losses, i.e, losses due to thermal expansion of existing vapors, vapor expansion caused by barometric pressure changes, and/or increases in the amount of vapor due to increased volatilization in the absence of a liquid-level change, can be accomplished via an empirical correlation that was developed by the American Petroleum Institute and that is presented in Reference 5. Necessary assumptions and observations include:

- Molecular weight of vapor = 4 x 29 = 116
- Vapor pressure of liquid = 80 mm Hg = 1.55 psia
- Tank diameter = 35 ft = 10.7 m
- Average vapor space height = 15 ft = 4.6 m
- Day/night average temperature change = 20°F = 11°C
- Light gray tank roof and shell in good condition (paint factor of 1.33).

With these assumptions, breathing losses would be estimated to be on the order of 30 kg per day in the absence of an emission control system. However, storage tanks at Emelle do have such systems and it is necessary to account for their efficiency in controlling airborne emissions.

CWM personnel (10) have indicated that carbon filters on tank vents are replaced when efficiency drops to 90 percent or below. They did not, however, have any estimate of the efficiency of new filters, thus complicating determination of the average efficiency of filters in service. It is therefore somewhat arbitrarily assumed that this average efficiency would be on the order of 94 percent. Thus, breathing losses from a single tank are estimated as 1.8 kg per day or 660 kg per year. Losses from four tanks are estimated to be 7.3 kg per day or 2650 kg per year.

As in the prior case, Reference 5 also includes an empirical correlation for working losses from fixed-roof tanks. Using similar assumptions, it is estimated that these losses would be on the order of 1300 kg per year (all four tanks) for support of 14 ocean voyages and 1700 kg per year in support of 18 ocean voyages of an incinerator ship. The overall totals for all losses are estimated to be 4000 kg per year for 14 voyages and 4400 kg per year for 18 voyages.

It is readily apparent that the storage tank losses predicted for Emelle are almost an order of magnitude greater than the analogous losses predicted from storage tanks installed at Chickasaw. It is well, therefore, to briefly comment on the reasons for this wide discrepancy.

The primary differences between the two sites involve preferences vis-a-vis the number and size of storage tanks. CWM would likely install four tanks of a fixed-roof design at Emelle, whereas OCS personnel (7) prefer two tanks of a covered floating-roof design for installation at Chickasaw. Given comparable emission control systems, with Emelle acutally being given an advantage in this area based on discussions with CWM personnel, fixed-roof tanks evolve considerably more emissions than covered floating-roof tanks. Additionally, it is apparent that four smaller tanks have a higher total emission rate than two tanks of equal capacity. It is stressed that these results would change dramatically if CWM decided to install covered floating-roof tanks at Emelle instead of using new or existing fixed-roof tanks to support future ocean incineration operations.

The accuracy of the rough order of magnitude emission rate estimate is further supported by data presented in Table 8, extracted verbatim from Reference 5. These data suggest substantially higher emission rates for fixed-roof tanks than floating-roof tanks.

Truck Loading Emissions

It has been estimated that 151 truck loads would be required to transfer wastes from Emelle to Chickasaw to support each of 14 voyages of an incinerator ship. Due to differences in waste characteristics, 163 truck loads would be required per voyage in the event of 18 trips.

Tank trucks and trailers are top loaded at Emelle. The loading bays are equipped with a vapor recovery system that transfers displaced vapors back to storage tanks.

According to the EPA (5), control efficiencies for vapor recovery systems range from 90 to 98 percent depending on the nature of the vapors and the type of recovery equipment employed. These efficiencies can be directly applied to estimates of emissions from tank loading operations.

Actual estimation of loading losses via an empirical correlation presented in Reference 5 requires the following assumptions.

- Molecular weight of vapors = 4 x 29 = 116
- Vapor pressure of liquid = 80 mm Hg = 1.55 psia
- Bulk temperature of liquid = 70°F = 530°R
- Saturation factor = 0.60 for submerged loading in normal dedicated service, and
- Control system efficiency = 98%.

The emission control system efficiency is taken to be at the upper end of the range since vapors are directly returned to storage tanks. EPA discusses the range in the context of systems that recover hydrocarbon vapors via refrigeration, absorption, adsorption, and/or compression. The system at Emelle simply contains contaminated air

TABLE 8

EVAPORATIVE EMISSION FACTORS FOR STORAGE TANKS WITHOUT CONTROLS

Product Stored	Fixed roof tanks						Floating roof tanks						Variable vapor space tanks	
	Breathing loss				Working loss		Standing storage loss				Withdrawal loss		10,500 bbl	
	"New tank" conditions		"Old tank" conditions				"New tank" conditions		"Old tank" conditions				Filling loss	
	lb/day-10^3 gal	kg/day-10^3 liters	lb/day-10^3 gal	kg/day-10^3 liters	lb/10^3 gal throughput	kg/10^3 liters throughput	lb/day-10^3 gal	kg/day-10^3 liters	lb/day-10^3 gal	kg/day-10^3 liters	lb/10^3 gal throughput	kg/10^3 liters throughput	lb/10^3 gal throughput	kg/10^3 liters throughput
Fuels - 67,000 bbl tanks														
1 Gasoline RVP 13	0.30	0.036	0.34	0.041	10.0	1.2	0.044	0.0052	0.10	0.012	0.023	0.0028	9.6	1.2
2 Gasoline RVP 10	0.23	0.028	0.26	0.031	8.2	0.99	0.033	0.0040	0.078	0.0094	0.023	0.0028	7.7	0.93
3 Gasoline RVP 7	0.16	0.019	0.18	0.022	5.7	0.68	0.023	0.0028	0.055	0.0066	0.023	0.0028	5.4	0.65
4 Crude oil RVP 5	0.064	0.0077	0.073	0.0088	2.8	0.34	0.012	0.0014	0.028	0.0034			Not used	Not used
5 Jet naphtha (JP 4)	0.086	0.010	0.098	0.011	2.5	0.30	0.012	0.0014	0.028	0.0034			2.3	0.28
6 Jet kerosene	0.0043	0.00052	0.0049	0.00059	0.027	0.0032	0.00054	0.000065	0.0013	0.00016			0.025	0.0030
7 Distillate fuel no. 2	0.0039	0.00047	0.0044	0.00053	0.023	0.0028	0.00049	0.000058	0.0011	0.00014			0.022	0.0026
8 Residual oil no. 6	0.00016	0.000019	0.00018	0.000022	0.00018	0.000022	0.000018	0.0000022	0.000043	0.0000052			0.00017	0.000020
Fuels - 250,000 bbl tanks														
9 Gasoline RVP 13	0.22	0.026	0.25	0.030	10.0	1.2	0.025	0.0030	0.057	0.0068	0.013	0.0015	Not used	Not used
10 Gasoline RVP 10	0.17	0.020	0.19	0.023	8.2	0.95	0.019	0.0023	0.044	0.0053	0.013	0.0015	Not used	Not used
11 Gasoline RVP 7	0.12	0.014	0.13	0.016	5.7	0.68	0.013	0.0016	0.031	0.0037	0.013	0.0015	Not used	Not used
12 Crude oil RVP 5	0.046	0.0055	0.052	0.0062	2.8	0.34	0.0077	0.0092	0.018	0.0022			Not used	Not used
13 Jet naphtha (JP 4)	0.062	0.0074	0.071	0.0085	2.5	0.30	0.0068	0.00082	0.016	0.0019			Not used	Not used
14 Jet kerosene	0.0031	0.00037	0.0035	0.00042	0.027	0.0032	0.00031	0.000037	0.00074	0.000089			Not used	Not used
15 Distillate fuel no. 2	0.0028	0.00034	0.0032	0.00038	0.023	0.0028	0.00028	0.000034	0.00068	0.000082			Not used	Not used
16 Residual fuel no. 6	0.00012	0.000014	0.00014	0.000017	0.00018	0.000022	0.000010	0.0000012	0.000024	0.0000029			Not used	Not used
Petrochemicals[a] - 67,000 bbl tanks														
17 Acetone	0.12	0.014	0.14	0.016	4.0	0.48	0.017	0.0020	0.039	0.0047			3.8	0.45
18 Acrylonitrile	0.060	0.0072	0.068	0.0082	1.8	0.21	0.0084	0.0010	0.020	0.0024			1.7	0.20
19 Benzene	0.079	0.0094	0.090	0.011	2.2	0.27	0.011	0.0013	0.026	0.0031			2.1	0.25
20 Carbon disulfide	0.24	0.029	0.28	0.033	8.8	1.1	0.035	0.0042	0.083	0.0099			8.2	0.98
21 Carbon tetrachloride	0.17	0.021	0.20	0.024	5.2	0.62	0.024	0.0029	0.056	0.0069			4.8	0.58
22 Chloroform	0.21	0.025	0.24	0.029	7.1	0.86	0.030	0.0036	0.071	0.0085			6.7	0.80
23 Cyclohexane	0.085	0.010	0.096	0.012	2.4	0.29	0.012	0.0014	0.028	0.0034			2.3	0.27
24 1,2 Dichloroethane	0.087	0.010	0.10	0.012	2.4	0.28	0.012	0.0014	0.029	0.0034			2.2	0.27
25 Ethyl acetate	0.083	0.010	0.095	0.011	2.3	0.28	0.012	0.0014	0.027	0.0033			2.2	0.26
26 Ethyl alcohol	0.028	0.0034	0.032	0.0038	0.66	0.079	0.0039	0.00046	0.0091	0.0011			0.62	0.074
27 Isopropyl alcohol	0.031	0.0038	0.036	0.0043	0.72	0.086	0.0043	0.00052	0.010	0.0012			0.68	0.082
28 Methyl alcohol	0.036	0.0044	0.042	0.0050	1.1	0.13	0.0051	0.00061	0.012	0.0014			1.0	0.12
29 Methylene chloride	0.31	0.037	0.35	0.042	11.0	1.3	0.044	0.0053	0.10	0.012			10.0	1.2
30 Methyl ethyl ketone	0.073	0.0087	0.083	0.0099	2.1	0.25	0.010	0.0012	0.024	0.0029			1.9	0.23
31 Methyl methacrylate	0.038	0.0046	0.043	0.0052	0.72	0.086	0.0051	0.00061	0.012	0.0015			0.68	0.082
32 1,1,1 Trichloroethane	0.17	0.020	0.19	0.023	5.1	0.61	0.023	0.0028	0.055	0.0066			4.8	0.58
33 Trichloroethylene	0.11	0.013	0.12	0.014	2.8	0.34	0.015	0.0018	0.035	0.0042			2.6	0.31
34 Toluene	0.035	0.0042	0.040	0.0048	0.66	0.079	0.0048	0.00058	0.011	0.0014			0.62	0.074
35 Vinyl acetate	0.092	0.011	0.10	0.013	2.7	0.32	0.013	0.0016	0.030	0.0037			2.5	0.30

[a] Due to safety and health considerations, toxicity, and value of these petrochemicals, they are normally stored in tanks with vapor recovery controls which are 90 to 98 percent efficient.

Emission factors based on the following parameters:

Ambient conditions:
Average temperature 60°F (15.6°C)
Diurnal ambient temperature change 15°F (8.3°C)
Wind velocity 10 mi/hr (4.5 m/sec)

Typical fixed roof tanks:
Outage 50 percent of tank height
Turnovers per year (N) 30 for crude, 13 for all other liquids
Paint factor (F_p) New tank white paint 1.00
Old tank white/aluminum paint 1.14

For 67,000 bbl tankage (10.7 x 10^6 liters)
Height 48 ft (14.6m)
Diameter 110 ft (33.5m)

For 250,000 bbl tankage (39.7 x 10^6 liters)
Height 44 ft (13.4m)
Diameter 200 ft (60.8m)

Emission factors based on the following parameters:

Typical floating roof tanks:
Paint factor (K_p) New tank white paint 0.90
Old tank white/aluminum paint 0.95
Seal factor (K_s) New tank modern seals 1.00
Old tank 50 percent old seals 1.14
Tank type factor (K_t) New tank welded 0.045
Old tank 50 percent riveted 0.068

Typical variable vapor space tank:
Diameter 50 ft (15.2m)
Height 30 ft (9.1m)
Capacity 10,500 bbl (1.67 x 10^6 liters)
Turnovers per year (N) 6
Volume expansion capacity = one fourth of liquid capacity 2625 bbl (0.42 x 10^6 liters)

volumes and is therefore likely to compare very favorably with these methods.

Given the above assumptions, it is estimated that a total of 570 kg of emissions might escape to the atmosphere during the loading of 2114 trucks in support of 14 ocean voyages of an incinerator ship. For 18 ocean voyages, the total is 730 kg from 2934 truck loading operations.

B.3 ACUTE SPILL EVENTS

B.3.1 Introduction

This section addresses the frequency and magnitude of spills at each of the facilities involved in the first three scenarios. The first discussion is on expected spill quantities, the next focuses on frequencies, and then three sections covering the present Chickasaw facility, the modified Chickasaw facility, and the portion of Emelle which supports Chickasaw follow.

B.3.2 Spill Sizes

Comprehensive assessment of the hazards/risks associated with a waste disposal facility requires consideration of spill size distributions associated with bulk cargo transfer operations. The following discussion addresses these activities for bulk cargo carriers (i.e., tank ships and barges) and highway tank vehicles (i.e., tank trucks and trailers, referred to in aggregate herein as "tank trucks").

In both cases considered, it was found that attention in past studies has been concentrated upon spills occurring while vehicles or vessels are in transit. Thus, there is little information available on transfer operations that can be considered current and valid. Fortunately, however, both the U.S. Coast Guard and the Materials Transportation Bureau of the U.S. Department of Transportation maintain data bases that are accessible and that provide opportunities for obtaining desired information in the future.

Spills Involving Ships and Barges

For over a decade, the U.S. Coast Guard has maintained the Pollution Incident Reporting System (PIRS), this being a computerized data base of oil and non-oil spills into water that have been reported to the Coast Guard. Table 9 provides summary data from this system, as published (11) annually by the Coast Guard, for incidents involving bulk cargo transfer operations at onshore or offshore marine facilities. Most, if not all, of these incidents probably involve cargo transfer to or from tank ships or tank barges.

An analysis of spill size distributions was undertaken in a study (12) for the EPA in 1978 to determine non-oil spill frequencies and size distributions. The data sources used were a listing of non-oil spills reported to the EPA prior to August of 1975 and similar spill reports received by the Pollution Incident Reporting System (PIRS) for calendar year 1976. Table 10 presents the resulting spill size distribution for spills attributed to the loading and unloading of vessels at marine terminals. The average spill volume for spills in excess of 100 gallons was determined to be 7,835 gallons.

Another source of data is the Oil Spill Intelligence Report (OSIR), an international weekly newsletter (13) that provides an annual listing of oil spills throughout the world involving spillage of 10,000 gallons or more of oil. Table 11 lists those events occurring during the three-year period 1980 to 1982 that were reported to involve vessel loading/unloading operations and had known oil spill volumes. These data can serve to better define the upper bounds of

TABLE 9

USCG STATISTICS FOR TRANSFER SPILLS AT MARINE FACILITIES

Year	No. Incidents	Spill Volume (gallons)
1982	270	286,980
1981	332	454,655
	602	741, 635

Average spill (1981-82) = 1,232 gallons

Year	No. Incidents	Spill Volume (gallons)
1977	497	500,384
1976	321	333,712
1975	276	92,522
1974	367	1,286,289
1973	271	309,141
	1,732	2,522,048

Average spill (1973-77) = 1,456 gallons

Source: Reference 11

TABLE 10

EPA/PIRS CHEMICAL SPILL DATA FOR LOADING/UNLOADING VESSELS

Spill Volume (gallons)	No. Spills	Percent of Total Number
1-100	66	68.04
101-500	19	19.59
501-1,000	2	2.06
1,001-5,000	4	4.12
5,001-10,000	2	2.06
10,001-50,000	2	2.06
>50,000	2	2.06
	97	99.99

Note: Only unloading/loading spills are considered
Source: Reference 12

TABLE 11

OSIR OIL SPILL DATA FOR TRANSFER OPERATIONS

Date	Location	Spill Volume (gallons)	Comments
7 Mar. 80	Texas	10,000	Line separation - transfer mishap
16 Apr. 80	Texas	10,000	Tank rupture - transfer mishap
8 Oct. 80	Alabama	36,000	Loading mishap - overflow
2 Nov. 80	N.J.	20,000	Loading mishap - overflow
30 Nov. 80	Texas	36,000	Loading mishap - overflow
18 Dec. 80	Maine	10,000	Offloading - defective valves
24 Dec. 80	Texas	138,000	Loading mishap - overflow
25 Jan. 81	U.K.	10,000	Broken loading arm
28 Feb. 81	Brazil	62,000	Offloading mishap
15 Aug. 81	Texas	17,000	Open valve during fueling
10 Sep. 81	France	40,000	Loading spill
10 Jul. 82	Texas	16,000	Cargo transfer mishap
14 Sep. 82	Canada	26,000	Barge unloading mishap
17 Sep. 82	New York	25,000	Hose break - barge loading
4 Dec. 82	U.K.	30,000	Transfer mishap

Note: Table only includes spills of 10,000 or more gallons of oil during cargo transfer operations

Source: Reference 13

spill volumes historically associated with high volume transfer operations.

Spills Involving Overflow of Receiving Tanks

One common result of tank loading/unloading accidents is an overflow of the receiving vessel. During 1981 and 1982, PIRS (11) recorded a total of 1,724 such incidents (all transportation modes and facilities) with an associated total spillage of 490,560 gallons of oil. The average spill was approximately 285 gallons. Averages for other general causes of oil spillage that may have been related to transfer operations included:

- Valve failures - 642 gallons
- Pump failures - 215 gallons
- Improper equipment operation - 387 gallons
- Other personnel error - 451 gallons.

Summary

As a result of the above, spill volumes for various components or operations have been determined. Valve failures (not small leaks which are considered as fugitive emissions) will be considered to yield 640 gallons (2,400 liters). Pump failures, again not including small releases through seals, are taken to yield 215 gallons (815 liters). Tank truck loading or unloading spills were found to be in the range of 300 gallons (1135 liters) on average. As the next section will detail some particular components which might fail, this quantity will be associated with loading or unloading arm spills. Finally, for ship transfer operations an average of about 1300 gallons (4920 liters) was found in Table 9. This would be the quantity associated with transfer piping or loading hose failures.

Spill quantities were also given for overfills, improper equipment operation and other personnel errors. The latter two categories are less than the valve, loading hose and pipeline failure quantities, and are therefore considered to be included in with the other estimates. In addition, as all the facilities in question have spill containment areas which can hold much larger quantities than the expected spill sizes discussed above, the accuracy of the predicted spill size need not be very great.

Overfill scenarios will not be very likely when unloading tank trucks, as all the storage tanks and ship tanks will have instrumentation and alarms indicating the liquid level, along with shutdowns on the ship's tanks. (Many older gasoline and oil storage tanks do not have such controls.) Moreover, on the storage tanks the chance of a particular truck being the one to exceed the tank's volume is relatively unlikely. For those scenarios where the tank truck is being filled, overfills are more likely. The discussion above indicates an expected spill size of 285 gallons (1080 liters).

For storage tanks, the size of the tank which fails is an important factor. As the tanks will be in spill containment areas and small leaks will be readily detectable, only catastrophic failures will be included in this analysis. The time-averaged volume of the tank will be 50% or less, so 50% of the tank volume will be assumed as the spill quantity.

B.3.3 Release Frequencies

In addition to looking at the expected spill size or spill size distribution associated with bulk cargo transfer operations, it is also necessary to consider the frequency of such spills. This section covers the types of components associated with the existing facility, an improved transfer facility incorporating interim storage facilities, and Emelle.

Without conducting a full risk analysis complete with fault tree development, it would seem that the primary cause of leaks and spills would be from leaking equipment or broken lines and hoses. The approach taken in this section is to attempt to develop estimates of these frequencies based on historical data and then to also discuss some potential scenarios involving human errors.

Frequencies of Failure

Most failure rate data provide information on failure to function without specifying whether leakage is occurring, particularly for valves. However, Wash-1400 (Reference 14) does give rates for leakage as well as plugging or staying open for many types of valves. Table 12 gives some rates from Wash-1400 which are relevant to the facilities in question. These data are based on experience from a number of industries, not just the nuclear industry. The table gives a median failure rate and the corresponding upper and lower bounds.

Unfortunately, the median represents a nonconservative estimate of the likelihood of failure and does not have the same additive and multiplicative properties as does the mean. Based on Rasmussen's assumption that the true failure rate is lognormally distributed, the value of the mean can be determined from the median and the estimated bounds. If there is only one order of magnitude of difference between the bounds, the mean is approximately equal to the median; with a range of two orders of magnitude the mean is about 3 times greater than the median; with a range of three orders of magnitude, the mean is about 9 times greater than the median.

Table 13 gives additional failure rates, derived from a number of other sources. Some of these give further data on the same components as in Table 12, others are for pieces of equipment not covered by Table 12. Pump failure rates are usually for stream stoppage, not leaks. Most pump leaks will be through gaskets and seals and are counted as fugitive emissions. Larger releases might be due to cracked casings or affiliated valves and piping. The valve and piping failures are included separately, and a proprietary data source estimates casing cracks as 3% of all failures. Storage tanks can have leakage rates that run the gamut from once per year to virtually no leaks ever, depending on the size of leak in question, the tank construction and material, and

TABLE 12

FAILURE RATES (WASH-1400)

Component	Failure Mode	Median Failure Rate	Range (Bounds)
air operated valve	fail to operate	3×10^{-4}/demand	$10^{-4} - 10^{-3}$
	external leak-rupture	10^{-8}/hour	$10^{-9} - 10^{-7}$
check valve	external leak-rupture	10^{-8}/hour	$10^{-9} - 10^{-7}$
relief valves	premature open	10^{-5}/hour	$3 \times 10^{-6} - 3 \times 10^{-5}$
pipes >3"	rupture^{+}	10^{-10}/hour	$3 \times 10^{-12} - 3 \times 10^{-9}$
pipes <3"	rupture^{+}	10^{-9}/hour	$3 \times 10^{-11} - 3 \times 10^{-8}$
gaskets	leak	3×10^{-6}/hour	$10^{-7} - 10^{-4}$
flanges, closures, elbows	leak/rupture	3×10^{-7}/hour	$10^{-8} - 10^{-5}$
welds	leak	3×10^{-9}/hour	$10^{-10} - 10^{-7}$
manual valve	rupture	10^{-8}/hour	$10^{-9} - 10^{-7}$
pumps	fail to run	3×10^{-5}/hour	$3 \times 10^{-6} - 3 \times 10^{-4}$

$^{+}$per section - usually defined as a segment between major discontinuities like valves, pumps, etc. Each section can contain several welds and flanges.

Source: Reference 14.

TABLE 13

ADDITIONAL FAILURE RATES

Component	Failure Mode	Failure Rate
+ Piping <3"	significant leakage	~10^{-7}/hour (per section)
+ Piping ≥3"	significant leakage	~3×10^{-8}/hour (per section)
+ Hoses - heavily stressed	rupture	4×10^{-5}/hour
- lightly stressed	rupture	4×10^{-6}/hour
+ Loading arms	rupture	3×10^{-8}/hour
	leakage	3×10^{-6}/hour
/ Sludge Tank	leakage	5×10^{-1}/year
+ Atmospheric Storage Tanks	serious leakage	10^{-4}/year
	catastrophic rupture	6×10^{-6}/year
o Hydrocarbon fuel hose	severe duty - all	6×10^{-1}/year
+ Liquid Quick Disconnect	all	4×10^{-7}/hour
* Metal Storage Tank	all	2×10^{-6}/hour
* Butterfly valve	all	2×10^{-7}/hour
* General Hose	all	2×10^{-6}/hour

Sources

+ Reference 15
* Reference 16
o Proprietary
/ Reference 17

the tank contents. This study is concerned with major losses of tank contents, as from ruptures or serious leakage.

In addition to the failure rates in the tables there are a number of other failure modes which might give rise to a release. These include:

- external events such as hurricanes, flooding, aircraft impact, etc.
- operator errors such as driving away with loading arms connected, opening drain valves prematurely, blocking in pumps, collisions with pipebridge, etc.

The potential for each of these events exists at Chickasaw and Emelle and their chances of occurrence will depend on design parameters in the event of external events and adequate procedures, supervision and training in the case of operator induced failures. The presence of quick disconnects and check valves should limit the release in the event of a driver pull-away. Various sensing mechanisms with shutoff switches are also available to limit releases. Such pullaways have been estimated to occur in one of every ten thousand or so loading/unloading operations. The actual rate will depend on the specific conditions at a given facility.

Crashes into the pipelines can probably be considered to already be incorporated in the failure rates, especially since the facilities have very little need for large vehicles on-site except for the tank trucks.

Overfilling of the ship's cargo tanks is also possible, but careful attention to the maintenance of the various level alarms and indicators as well as well-specified procedures for cargo tank selection should help limit the possibility. Incorrect hose connections are usually estimated at about 5×10^{-3}/operation (Reference 15), but pressure testing the loading arms and hose before beginning operations helps limit the consequences of such errors.

Summary

The release frequencies which will be used in this analysis are:

Valves (except relief)	3×10^{-8}/hr
Relief valves	10^{-5}/hr
Pumps	3×10^{-6}/hr
Loading hose (lightly stressed)	4×10^{-6}/hr
[+]Unloading arms	3×10^{-6}/hr
[+]Pipelines <3"	10^{-7}/hr (per section)
>3"	3×10^{-8}/hr (per section)
[*]Storage tanks	10^{-4}/yr

Some of these rates are for major leaks while others are for ruptures or catastrophic failures.

B.3.4 Existing Chickasaw Facility

Facility Description

The current system is composed of the following elements:

- 4 unloading arms, each of which has a
  - swivel joint to common header
  - 4" butterfly valve for isolation
  - 4" dry disconnect
- common header with check valve to avoid backflow into trucks
- 2 400 gpm positive displacement pumps, each of which has a
  - 4" butterfly valve for isolation on intake and discharge lines

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[+] Significant leakage

[*] Serious Leakage

  - pressure relief valve (to recirculation) on the discharge line
  - pressure switch (low)

- 2" recycle line from one pump with one valve
- 750' aboveground pipeline composed primarily of 8" pipe, ending in a 6" section, involving one pipebridge across roadway
- fail safe pneumatic operated 6" valve which closes within 30 seconds
- zero flow shutdown switch and high pressure switch on pipeline
- 50' long 6" flexible hose to connect to ship
- isolation valves at each end of hose
- drainage system composed of small valves on
  - each pump (3/4")
  - common header (several valves, but only one connects to header) (1")
  - pipeline after pipebridge (1")
  - loading platform (2")
- numerous 3/4" air vents and flanges

Most of these items are potential points of release.

At a loading rate of 800 gpm, it takes almost twenty hours of pumping to load the ship. However, it is not conceivable that four trucks will always be available for continuous filling, plus there is necessarily time required for hooking up the trucks and the incinerator ship initially. It has been estimated (Reference 18) that activities associated with loading are underway for 7.5 days at a time, 14 or 18 times a year for the current system.

Spill Frequencies

The applicability of the rates in Tables 13 and 14 to the Chickasaw facility depends on the comparability of equipment to that for which the data was collected in terms of operating environment, preventive maintenance, size, severity of duty, etc. However, the rates can be applied to the equipment listed in the previous section in order to get an approximation of the overall spill frequencies. Small air vents, flanges and drain lines are not included. As per Reference 18, an exposure time of 180 hours is used.

- Valves: There are a total of about 16 butterfly valves of concern - one per unloading arm, two per pump, two for the loading hose, one on the recycle line, and five primary drain valves. There are also one check valve and two relief valves. The relief valves will release to the recirculation line and are not included in the total. There is one air-operated safety valve. For 14 voyages per year, the results are:

  air-operated - 3×10^{-8}/hour x 180 hr x 14/yr = 7.6×10^{-5}/yr

  check - 3×10^{-8}/hr x 180 hrs x 14/yr = 7.6×10^{-5}/yr

  manual - $16 \times 3 \times 10^{-8}$/hr x 180 hrs x 14/yr = $\underline{1.2 \times 10^{-3}/\text{yr}}$

  1.4×10^{-3}/yr

  If there are 18 voyages per year, the total becomes 1.7×10^{-3}/yr, again excluding relief valves.

- Loading Hose: There is one flexible hose. The fact that it is not used continuously at its full rate makes the lightly stressed rate most appropriate. Pinhole leaks are probably not covered by this rate. The results for 14 voyages per year are:

  - 4×10^{-6}/hr x 180 hrs x 14/yr = 10^{-2}/yr

  For 18 voyages per year this becomes 1.3×10^{-2}/yr.

- Unloading Arms: There are four arms, each with a dry disconnect. The leakage rate for the loading arm alone has been used, although there could be a contribution from the swivel joint and the disconnect. For 14 voyages per year the result is:

- $4 \times 3 \times 10^{-6}$/hr x 180 hrs x 14/yr = 3×10^{-2}/yr

For 18 voyages per year this changes to 3.9×10^{-2}/yr.

- Pipelines: There are the common header, the recycle line, and the transfer pipeline. Small drainage lines are not of as much concern. The primary pipeline is taken as ten sections - based on length more than discontinuities (includes piping around pumps), the common header as two, and the recycle line as one. For 14 voyages per year the results are:

  $> 3''$ - $12 \times 3 \times 10^{-8}$/hr x 180 hrs x 14/yr = 9.1×10^{-4}/yr

  $\leq 3''$ - 1×10^{-7}/yr x 180 hrs x 14/yr = $\underline{2.5 \times 10^{-4}/\text{yr}}$

  1.2×10^{-3}/yr

  If there are 18 voyages per year this becomes 1.5×10^{-3}/yr.

- Pumps: There are two pumps. For 14 voyages per year the result is:

  - $2 \times 3 \times 10^{-6}$/hr x 180 hrs x 14/yr = 1.5×10^{-2}/yr

  For 18 voyages per year this becomes 1.9×10^{-2}/yr

These estimates are sensitive to the number of hours each component is actually used. The full 180 hours per voyage is probably reasonable for the pipeline and loading hose, but not for each pump and each loading arm. The pipeline leakage rates are for fairly significant leaks - smaller leaks are expected to occur more frequently, and might be comparable to leaks from welds, flanges, valve stem packing, etc. The rates are also sensitive to an accurate breakdown into a number of sections.

B.3.5 Modified Chickasaw Facility

The improved facility at Chickasaw will have many of the same components as before, but will also include two 800,000 gallon storage tanks. Associated with these tanks will be additional valves, piping, and pumps. It has been assumed that there will be four additional pumps, approximately sixteen new butterfly valves, and ten additional sections of piping.

Ocean-Going Incineration Ship

The improved facility will have ship loading occur over a much shorter period of time - 1.5 days per voyage (Reference 18) - but tank truck unloading will probably occur more often at a lower daily throughput. This is due to the fact that it will no longer be necessary to have batch unloadings and tank trucks will not have to sit idle acting as interim storage facilities.

To handle this, it was decided to assume that half the piping and half the valves - basically those between the storage and truck unloading areas - would be exposed all the time, in that they would never be drained and decontaminated. The unloading pumps and unloading arms were assumed to be in operation for a maximum of 180 hours per voyage - where the same comment regarding the conservatism of this assumption applies. The remaining piping, valves, and new pumps were assumed to be exposed 36 hours per voyage.

The results of these assumptions are then, by component, given below.

- Valves: As in the previous section, relief valves on the pumps will not be included. The check valve is associated with the truck bays, and the safety valve with the ship loading area. For 14 voyages per year

  - $17 \times 3 \times 10^{-8}/\text{hr} \times 8760 \text{ hrs/yr} = 4.5 \times 10^{-3}/\text{yr}$
  - $17 \times 3 \times 10^{-8}/\text{hr} \times 36 \text{ hrs} \times 14/\text{yr} = \underline{2.6 \times 10^{-4}/\text{yr}}$

    $4.8 \times 10^{-3}/\text{yr}$

  For 18 voyages per year the same total is obtained.

- Pipelines: For 14 voyages per year, the results are:

  - $1 \times 10^{-7}/\text{hr} \times 8760 \text{ hrs/yr} = 8.8 \times 10^{-4}/\text{yr}$

- $12 \times 3 \times 10^{-8}$/hr x 8760 hrs/yr = 3.2×10^{-3}/yr
- $10 \times 3 \times 10^{-8}$/hr x 36 hrs x 14/yr = $\underline{1.5 \times 10^{-4}/yr}$

4.2×10^{-3}/yr

For 18 voyages the total becomes 4.3×10^{-3}/yr

- Pumps: Two pumps are exposed 180 hours per voyage and four are exposed 36 hours per voyage. This gives, for 14 voyages per year:
  - $2 \times 3 \times 10^{-6}$/hr x 180 hrs x 14/yr = 1.5×10^{-2}/yr
  - $4 \times 3 \times 10^{-6}$/hr x 36 hrs x 14/yr = $\underline{6 \times 10^{-3}/yr}$

  2.1×10^{-2}/yr

  For 18 voyages, this becomes 2.7×10^{-2}/yr.
- Unloading Arms: These results remain the same as for the current facility, 3×10^{-2}/yr for 14 voyages and 3.9×10^{-2}/yr for 18 voyages.
- Loading Hose: It is assumed that there continues to be one flexible hose. For 14 voyages per year the result is
  - 4×10^{-6}/hr x 36 hrs x 14/yr = 2×10^{-3}/yr

  For 18 voyages per year this becomes 2.6×10^{-3}/yr
- Storage Tanks: There are two tanks, each exposed for a full year, regardless of number of voyages. As a result, serious leakage or rupture is expected 2×10^{-4}/yr.

Land-Based Incinerator

If a land-based incinerator is to be used instead of an ocean-going ship, the exposure times are again changed, although roughly the same components are to be expected.

The unloading arms and pumps would probably be in use the same number of maximum hours (14 x 180 or 18 x 180), the valves and piping would all be exposed continuously, and the pumps to the incinerator would not all run simultaneously but might each run the same amount of time (36 x 14 or 36 x 18 hours apiece). This gives the results listed below.

- **Valves**: For either type of waste, the total would be
  - $34 \times 3 \times 10^{-8}$/hr x 8760 hrs/yr = 8.9×10^{-3}/yr
- **Pipelines**: For either type of waste, the total would be
  - 1×10^{-7}/hr x 8760 hrs/yr = 8.8×10^{-4}/yr
  - $22 \times 3 \times 10^{-8}$/hr x 8760 hrs/yr = $\underline{5.8 \times 10^{-3}}$/yr

    6.7×10^{-3}/yr
- **Pumps**: For the PCB waste, the results would be
  - $2 \times 3 \times 10^{-6}$/hr x 2520 hrs/yr = 1.5×10^{-2}/yr
  - $4 \times 3 \times 10^{-6}$/hr x 504 hrs/yr = $\underline{6 \times 10^{-3}}$/yr

    2.1×10^{-2}/yr

  For the EDC waste, the total is 2.7×10^{-2}/yr.
- **Unloading Arms**: For the PCB waste, the total would be
  - $4 \times 3 \times 10^{-6}$/hr x 2520 hrs/yr = 3×10^{-2}/yr

  For the EDC waste, the total becomes 3.9×10^{-2}/yr.
- **Storage Tanks**: For either type of waste the total is 2×10^{-4}/yr.

B.3.6 Emelle

As discussed in Section B.2.3, the facilities at Emelle which might be considered as support of Chickasaw include:

- truck unloading facilities similar to those at Chickasaw;
- truck loading facilities similar to the unloading ones;
- 4 200,000 gallon storage tanks;
- double-piped pipelines; and
- roughly 30 valves

It was decided to assume that there would not be serious leakage problems from the double-pipe, that the 30 valves would be continuously exposed, that the four unloading arms and two pumps would be in use a maximum of 180 hours per voyage, and that the same would be true for the loading arms and pumps.

The estimated exposure times for each pump and loading/unloading arm are highly approximate, but are felt to provide maximum upper bounds. Again, as spills will be contained, the accuracy of such numbers is not critical. Moreover, since dry disconnect and swivel joint failures are not called out separately, the higher exposure times help capture releases from these sources as well.

The results for Emelle are given below.

- Valves: For either type of waste the total is
  - $30 \times 3 \times 10^{-8}$/hr x 8760 hrs/yr = 7.9×10^{-3}/yr
- Pumps: For 14 voyages per year, the total is
  - $4 \times 3 \times 10^{-6}$/hr x 180 hrs x 14/yr = 3×10^{-2}/yr

  For 18 voyages this becomes 3.9×10^{-2}/yr
- Storage Tanks: For either waste type, the total is 4×10^{-4}/yr.
- Loading/Unloading Arms: For 14 voyages per year, the total is:
  - $8 \times 3 \times 10^{-6}$/hr x 180 hrs x 14/yr = 6×10^{-2}/yr

  For 18 voyages this becomes 7.8×10^{-2}/yr.

It may be the case that some trucks would bypass Emelle and go straight to Chickasaw while the ship was there, but the worst case has them all go via Emelle.

B.4 EVALUATION OF SCENARIOS

This section pulls together the results of Sections B.2 and B.3 for each of the alternate scenarios and for each waste type. Each scenario is discussed individually.

B.4.1 Ocean-Going Incinerator Ship with Storage at Emelle

Under this scenario, fugitive emissions and acute spills from both Emelle and the existing facilities at Chickasaw must be considered. Taking the totals from the relevant sections, the fugitive emissions are:

	PCB Waste (14 Voyages)	EDC Waste (14 Voyages)
at Emelle	4697 kg/yr	5269 kg/yr
at Chickasaw	163 kg/yr	209 kg/yr

These are not totalled as they occur at two distinct geographic locations. The emissions listed above are comprised of volatile components of the liquid wastes and do not necessarily include PCB's or other nonvolatile materials.

For acute spills the following frequencies and spill quantities were obtained:

Component	Spill Size (gallons)	Frequency for PCB Waste	Frequency for EDC Waste
Valves	640	9.3×10^{-3}/yr	9.6×10^{-3}/yr
Pumps	215	4.5×10^{-2}/yr	5.8×10^{-2}/yr
Pipelines	1300	1.2×10^{-3}/yr	1.5×10^{-3}/yr
Loading Hose	1300	10^{-2}/yr	1.3×10^{-2}/yr
Loading/Unloading Arm	300	9×10^{-2}/yr	1.2×10^{-1}/yr
Storage Tanks	100,000	4×10^{-4}/yr	4×10^{-4}/yr

These have been totalled for both locations, as they are expected to be contained and cleaned up. If the products of frequency and spill size are taken and summed, the overall losses expected each year are 98 gallons for the PCB waste and 113 gallons for the EDC waste.

B.4.2 Ocean-Going Incinerator Ship with Storage at Chickasaw

All releases, both fugitive and acute, occur at Chickasaw in this scenario. The fugitive emission totals for each waste type are

	PCB Waste	EDC Waste
Fugitive emissions	669 kg/yr	719 kg/yr

Again, PCB's and other nonvolatile components are not necessarily included.

For acute spills, the following frequencies and spill quantities were obtained:

Component	Spill Size (gallons)	Frequency for PCB Waste	Frequency for EDC Waste
Valves	640	4.8×10^{-3}/yr	4.8×10^{-3}/yr
Pumps	215	2.1×10^{-2}/yr	2.7×10^{-2}/yr
Pipelines	1300	4.2×10^{-3}/yr	4.3×10^{-3}/yr
Loading Hose	1300	2×10^{-3}/yr	2.6×10^{-3}/yr
Loading/Unloading Arm	300	3×10^{-2}/yr	3.9×10^{-2}/yr
Storage Tanks	400,000	2×10^{-4}/yr	2×10^{-4}/yr

The overall expected loss each year, for each waste type, is 105 gallons for the PCB waste and 110 gallons for the EDC waste.

B.4.3 Storage at a Land-Based Incinerator

This scenario is similar to the one above so far as storage and transfer releases are concerned. The fugitive emissions (not associated with the incinerator itself) are the same, except that emissions from the vapor return system for the ship are not included.

	PCB Waste	EDC Waste
Fugitive emissions	564 kg/yr	584 kg/yr

Again, PCB's and other nonvolatile components are not necessarily included.

For the acute spills the scenario differ slightly, as shown below.

Component	Spill Size (gallons)	Frequency for PCB Waste	Frequency for EDC Waste
Valves	640	8.9×10^{-3}/yr	8.9×10^{-3}/yr
Pumps	215	2.1×10^{-2}/yr	2.7×10^{-2}/yr
Pipelines	1300	6.7×10^{-3}/yr	6.7×10^{-3}/yr
Loading/Unloading Arm	300	3×10^{-2}/yr	3.9×10^{-2}/yr
Storage Tanks	400,000	2×10^{-4}/yr	2×10^{-4}/yr

The overall expected loss each year, for each waste type, is 109 gallons for the PCB waste and 113 gallons for the EDC waste.

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Appendix C

MARINE TRANSPORTATION RELEASE ANALYSIS

I. INTRODUCTION

In this Appendix, estimates are developed of the probability that hazardous wastes are accidentally spilled into the marine environment during the operation of the VULCANUS (4,691 deadweight tons).[1] Such estimates cannot be based on the actual operating history of incineration ships because not enough incineration voyages have been made to establish a meaningful safety record. Consequently, the estimates are based on casualty and spill data assembled for other types of tank ships[2] adjusted to reflect our judgement of how the operating and design characteristics of the VULCANUS are likely to affect casualty and spill rates.

Our estimates provide an indication of likely casualty and spill rates for a VULCANUS-type incineration ship. They should not be viewed as precise measures of these rates because of uncertainties in assessing how the casualty and spill rates of incineration ships may differ from the rates observed for tank ships in general.

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[1] Since our estimates are for VULCANUS-type vessels, they may not reflect the spill rates for the containerized operations proposed by Seaburn and Environmental Ocean Services.

[2] The term "tank ship" refers to any ship which carries liquid commodities in bulk. As examples, tank ships include petroleum product tankers, crude oil carriers, chemical product carriers, liquefied flammable gas carriers (e.g., LPG/LNG), and combination carriers such as OBO's when they are engaged in the bulk liquid trade. It is also important to understand that some tank ships carry combinations of these liquid commodities such as petroleum products with petrochemicals or petrochemicals with non-petroleum based chemicals. Therefore, the tank ship designations are not necessarily precise.

Our analysis of the probability of spills has two components. First, it estimates the probability of the occurrence of a spill. Second, it provides a probability distribution of spill sizes. The estimate of the frequency of spills is developed for four casualty types (collisions (ship-to-ship impacts), groundings (ship impacts with the sea bottom), rammings (ship-to-non-ship (e.g., piers, bridges, jetties, floating debris, etc.) impacts), and non-impact (fires, explosions, structural failures, etc.) casualties) and for four general locations (pier and harbor, Mobile Bay, the coastal area, and the burn zone).

The remainder of this Appendix is organized as follows: Section II provides a brief review of the operating record of incineration ships in Europe; Section III develops estimates of spill rates for each of the four casualty types within each of the four locations; Section IV develops an estimate of the distribution of the size of the spills; and, Section V provides a summation and comparative analysis.

II. OPERATING HISTORY OF INCINERATION SHIPS

Hazardous wastes have been incinerated off the European coast in the North Sea since 1972. Both VULCANUS I and VULCANUS II have operated out of European ports. Hazardous wastes have also been incinerated by MATHIAS I, MATHIAS II, and MATHIAS III and VESTA.

VULCANUS I, VULCANUS II, and VESTA have made about 320 voyages and incinerated approximately 650,00 metric tons of hazardous wastes which includes several trial burns made by VULCANUS I and VULCANUS II. [3] Additional information on these ships is provided in Table 1. No casualties, such as collisions, groundings, rammings, or fires, have occurred, nor have there been any spills from loading these ships in port. [4]

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[3] Information on the number of voyages made by MATHIAS I, MATHIAS II, and MATHIAS III and the amount of wastes incinerated was not available. However, it is known that these ships have not been involved in any casualties to date.

[4] European authorities generally treat incineration ships similar to chemical tankers, as opposed to the U.S. approach that will impose special restrictions on the operation of such vessels. Incineration ships in Europe generally operate out of several ports, whereas under the current proposal, VULCANUS will operate only out of one port, thus allowing shipboard personnel to have greater familiarity with the single port of operation. Incineration ships in Europe typically utilize chemical tanker facilities for loading, whereas in Mobile, they will use a specially dedicated loading facility. Additionally, the European incineration ships operate in the North Sea, a body of water that is fairly shallow (i.e., the average depth is about 40 meters), that has a very heavy volume of marine traffic, and where fog and low visibility present navigational problems in the Spring and Fall.

Although incineration ship operations in Europe have established a good safety record, the number of voyages is too small to be used in directly estimating the probability of such ship casualties and spills. Based on these data, it may be said that there is a 90 percent chance that the overall casualty rate is less than 7 in 1,000 per voyage. However, the probability that the casualty rate per voyage is greater than 2 in 1,000 could be as much as 50 percent.

Therefore, all that can be concluded from the actual operating record of the European incineration ship operations is that there is no evidence that the overall operations of incineration ships or the physical incineration of hazardous wastes at sea constitutes a high risk activity. In order to more specifically estimate the probability of spills of an incineration ship operation, a much larger operating population and spill data base are required.

III. ESTIMATED SPILL RATES

1. General Approach. The limited operating history of incineration ships and the absence of any marine casualties involving such ships to date, necessitates the use of an alternate data base to estimate spill rates. However, most ship casualty data sources do not segregate spill events by type of casualty or by ship size. Accordingly, we have chosen to use a subset of worldwide tank ship data for this analysis. We believe that this selection is the most appropriate for the following reasons. First, the incineration ship, like all tank ships, is engaged in the bulk carriage of a liquid cargo. Tank ship cargos typically include crude oil, petroleum products, petrochemicals, liquefied gases, and chemicals, but may include more innocuous cargos such as vegetable oils, wine, and even water. All tank ships are common in that their cargo spaces consist of tanks suitable for the bulk carriage of a liquid. They may differ, however, in terms of the cargo containment system used such as integral versus independent tanks [5] or "single hull" versus "double hull". [6]

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[5] An integral tank is one that is formed by the ship's structure. An independent tank, on the other hand, is not in anyway formed by the ship's structure but rather, is placed within the structure.

[6] Cargo tanks may or may not extend transversely to the ship's full beam or vertically from the ship's bottom to the main deck. Where such tanks do extend to the ship's full beam and from the ship's bottom to the main deck, the ship is referred to as a single hull tanker. Sometimes, the cargo tank bottom is formed by the top of a double bottom or the outboard side of the cargo tank is formed by a longitudinal bulkhead set inboard from the ship's side structure (i.e., a double side) or both (i.e., a double hull).

Ordinarily, a tank ship is considered to be an oil tanker; they are by far, the most prevalent type of tank ship. In general, oil tankers carry crude oil or some petroleum product in a longitudinal series of integral cargo tanks that span the entire beam of the ship and that extend vertically, from the ship's bottom to the main deck; i.e., a conventional single hull, oil tanker. Typically, these oil tankers are provided with a single, direct geared, fixed-pitch propeller and are not outfitted with a bow thruster.

The second reason we believe that the selection of this data base is most appropriate is that it provides a sufficiently large data base which permits examinations of various tank ship design characteristics and operational practices for comparison with the incineration ship. Within the world tank ship fleet, it is probably fair to say that in terms of spill potential, a double hull tank ship with a deadweight of about 2,000 to 10,000 long tons and which basically operates in a coastal environment, most closely represents the incineration ship operation. However, the population of double hull tank ships is so small that spill estimates made from that very small portion of the spill data base would be statistically questionable.[7] The next best option is to use the subset of the spill data base for tank ships between 2,000 and 10,000 long tons, since they not only approximate the incineration ship in size, but also tend to be ships that more typically make shorter, coastal voyages as opposed to longer, transoceanic voyages.

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[7] Spill data even for double hull tank ships of all size classes (including chemical and gas carriers) are extremely small. However, in an indicative sense, they suggest spill rates on the order of 2 to 3 per 100,000 voyages which is approximately three to five times better than the overall spill rate for all tank ships taken as a whole throughout the world.

Accordingly, spill rates for those tank ships within a deadweight range of 2,000 to 10,000 long tons were developed from a proprietary, worldwide, tank ship accident and spill data base entitled **ECOTANK** which is held and maintained by ECO, Inc., of Annapolis, Maryland. This data base presently contains approximately 10,000 worldwide, tank ship accidents and over 1,000 tank ship spill events that occurred during the period from 1969 to 1982. Among other things, each file lists the type of casualty (i.e., collision, grounding, ramming, explosion, fire, etc.); data on the ship's characteristics including deadweight tonnage; the resulting extent of damage; the location of the incident both geographically and by operating environment or location (i.e., at a pier, in a harbor or similar type body of water, within an entranceway, in a coastal zone, or at sea); and, spill data.

Historical spill rates (spill event per voyage) for tank ships whose deadweight fall within the range of 2,000 long tons or greater but less than 10,000 long tons were developed from this data base for the three categories of impact-type casualties (collisions, groundings, and rammings) and for a single, non-impact category (i.e., explosion, fire, structural failure, or capsizing). Within each of these four categories, estimates were made of the effect of the double hull, the controllable pitch (CP) propeller with a bow thruster, and the U.S. Coast Guard (USCG) navigation requirements upon the historical spill rates for tank ships which ordinarily do not have a double hull, a controllable pitch propeller with a bow thruster and, ordinarily, are not subjected to the USCG navigation requirements within a port and contiguous areas. Such effects (as a percentage) are then integrated with the historical spill rates to give the expected spill rates for the incineration ship operation.

2. Spill Rates for Collisions. Line 1 of Table 2 gives the observed historical spill rates for collisions by the four locations and in total that were experienced by tank ships of 2,000 to 10,000 deadweight tons throughout the world.[8] The second, third, and fourth lines respectively give the estimated values of one minus the effectiveness of: (i) the double hull; (ii) the CP propeller/bow thruster; and, (iii) the USCG navigation requirements. We estimate that the double side portion of the double hull

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[8] These historically observed spill rates for tank ships between 2,000 and 10,000 deadweight tons are considered to be most applicable for this analysis since the number of spill events in this size range is sufficiently large to permit the projection of expected values. Additionally, the VULCANUS is 4,691 deadweight tons which falls near the midpoint of the selected size range. **ECOTANK** has five categories of casualty/spill locations; specifically, pier, harbor, entranceway to a harbor, coastal (within 50 nautical miles of land) and at sea (beyond 50 nautical miles of land). Correspondence for this incineration ship anaylsis is established by relating **ECOTANK** pier and harbor to Mobile pier and harbor, **ECOTANK** entranceway to Mobile Bay, **ECOTANK** coastal directly to coastal and **ECOTANK** sea to the burn zone.

in the pier and harbor location will reduce the number of collision spill events by about 20 percent (0.20).[9] Thus, it is expected that the collision spill rate for the pier and harbor location will be eighty percent (0.80) of the corresponding historical spill rate. In the Mobile Bay, coastal, and burn zone locations, we reduced that effectiveness by the ratio of the difference of the squares of the speed regimes or kinetic energy levels to account for the extent of side penetration likely to result from a collision in each location. We estimate the effectiveness of the CP propeller/bow thruster in reducing collision spill events at about 25 percent (0.25).[10] Its effectiveness is also reduced according to the differences in kinetic energy from one location to the next.

The USCG navigation requirements refer to the restrictions placed on incineration ship movements between the terminal along Chickasaw Creek Channel and the Entrance Channel to Mobile Bay. They include limiting operations to daylight hours only and when the visibility is two nautical miles or greater and imposing a moving safety zone that will be strictly controlled by a Coast Guard escort vessel to restrict any traffic from coming within 100 yards of the incineration ship. When strictly controlled, this has the effect of essentially prohibiting any other major marine traffic from meeting or overtaking the incineration ship within the 400-foot wide, Lower and Upper Reach Channels of Mobile Bay and precluding the occurrence of any traffic from the Intracoastal Waterway and Hollingers Island Channel to cross or enter the main shipping channel when the incineration ship is in the vicinity of those intersections within Mobile Bay. In the Pinto Island, Mobile Channel, Blakeley Island, and St. Louis Point Reaches of the channel above Mobile Bay to the Chickasaw Creek Channel, with average channel widths of approximately 600 feet, the effect of these navigation requirements is to allow meeting or overtaking situations to occur only when, in the opinion of the Coast Guard escort vessel, the situation provides a sufficient margin of safety to do so. (It should also be pointed out that as part of the USCG navigation requirements, the incineration ship will be required to be escorted by tugboats.)

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[9] Source: "Reports on Parts 1 and 2 of Study I, Segregated Ballast Aboard Large Crude Carriers and Product Tankers and Smaller Crude Carriers," Notes by the United States of America to IMCO, June, 1972/February, 1973.

[10] Source: Card et al, "Report to the President on an Evaluation of Devices and Techniques to Improve Maneuvering and Stopping Abilities of Large Tankers," U.S. Coast Guard, Washington, D.C., September, 1979.

We estimate the USCG navigation requirements would reduce collisions by approximately 60 percent (0.60) in the pier and harbor location and by about 90 percent (0.90) in the Mobile Bay location. The estimates of 60 and 90 percent reductions for the effectiveness of the USCG navigation requirements have been derived from the distribution of collision events within the two locations according to three collision types. The first type consists of those collisions that occur between two underway vessels in meeting, overtaking, and crossing situations. The second type consists of those collisions that involve only one underway vessel with a second vessel that is either anchored or moored and the third type consists of those collisions that involve only one underway vessel with another stationary vessel during a docking or undocking process. In the pier and harbor location, no collision events during docking are considered to be possible since the incineration ship will be operating at a protected facility. Collisions between the incineration ship and other vessels along the channel are estimated to be reduced by 60 percent based upon the distribution of collision types within these locations. In the Mobile Bay location, since there are neither docking operations nor any moorings/anchorages, these collision events are considered to be reduced by 90 percent based upon the distribution of collision types with these locations.

In the coastal and burn zone locations, the USCG navigation requirements will not be in force except for a general broadcast to mariners. Thus, we feel that the major effect of an incineration ship's design characteristics as well as the operating restrictions placed on the VULCANUS would be to reduce the spill rate from collisions in the pier, harbor and Mobile Bay, rather than the coastal or burn zone locations.

3. Spill Rates from Groundings. Table 3 develops the expected spill rates for groundings. In addition to the three factors used in the case of collisions, there is an added factor entitled, "U.S. Gulf Coast". The spill rates for groundings along the U.S. Gulf Coast are significantly less than for the worldwide rate due, primarily, to the soft bottom conditions.

This is reflected on Table 3 by adjusting spill rates by a factor of 0.31; i.e., the spill rate for the U.S. Gulf Coast on an overall basis is about 69 percent (0.69) less than the worldwide rate. Within the pier and harbor location, we estimate that the double bottom will be approximately 90 percent (0.90) effective in reducing spills from groundings.[11] That effectiveness will be reduced by the difference in kinetic energy from one location to the next, since the extent of vertical penetration of a grounding will vary according to the expected kinetic energy level of the grounding which is a function of the square of the speed at the time of grounding. Thus, the downward adjustment in spill rates is less for the Bay and coastal area where the incineration ship would be moving at greater speeds. We estimate the effectiveness of the CP propeller and bow thruster in reducing spills from grounding events to be about 50 percent (0.50) at the pier and harbor location.[12] In the Mobile Bay location, that effectiveness is reduced to 25 percent, since at ship speeds of six knots or greater, the bow thrusters become less effective.[13] For the coastal and burn zone locations, the effectiveness of the CP propeller is further reduced according to the increased kinetic energy level or stopping distance. The USCG navigation requirements are estimated to reduce groundings by about 10 (0.10) percent in both the pier/harbor and Mobile Bay locations. [14] In the coastal and burn zone locations, these USCG navigation requirements are not applicable; therefore, there is no mitigating effect on groundings. Overall, the expected spill rate for groundings throughout all locations for the incineration ship operation is approximately 86 percent less than the historical rate for similar size tank ships throughout the world.

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[11] Source: Card, James C., "Effectiveness of Double Bottom in Preventing Oil Outflow from Tanker Bottom Damage Incidents," Marine Technology, January, 1978.

[12] Source: See footnote 9.

[13] Source: Porricelli, J.D., V.F. Keith and R.L. Storch, "Tankers and the Ecology," Transactions, Volume 79, Society of Naval Architects and Marine Engineers, New York, N.Y., 1971.

4. Spill Rates from Rammings. Table 4 develops the expected spill rates for rammings (ship to non-ship impacts). The adjustments or modifications used in each of the four locations for the double hull, the CP propeller/bow thruster, and USCG navigation requirements are identical to those used for groundings in Table 3.

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[14] The effect of the USCG regulated navigation area on impact type casualties other than collisions (i.e., groundings and rammings) is estimated to be minimal. In fact, a paper, "Collisions and Groundings: Preventing the Preventable," U.S. Naval Institute Proceedings, Annapolis, Maryland, December, 1974, by Captain R.C. Hill, USCG, estimates that the effect of such a regulated navigation area on non-collision impact casualties is on the order of a 10 percent decrease.

5. Spill Rates from Non-Impact Casualties. Table 5 gives the expected spill rates for non-impact casualties. Non-impact casualties are all casualty types except collisions, groundings, and rammings. They include such events as explosions, fires, structural failures, mechanical breakdowns and capsizings. Non-impact casualties occurring within the confines of the port are most likely to occur when the ship is moored at the pier, rather than while the ship is underway in the harbor.

We have elected not to adjust these spill rates to account for the features of the operation of the VULCANUS. This is because there is insufficient data on the frequency of specific types of non-impact casualties and because of the difficulty of estimating how the design and operation of the VULCANUS would affect the frequency of these types of casualties.[15] Some of the design characteristics and features of the operation of the VULCANUS that we think would reduce non-impact casualty and spill rates include:

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[15] With the exception of studies on the effects of cargo tank inert gas systems on the frequency of fires and explosions, little research has been conducted on the effectiveness of measures intended to mitigate the occurrence of non-impact casualties for tank ships.

- the double hull construction in the sense of added structural strength;

- the higher degree of intact and damage survivability required for this type vessel;

- the lower frequency of severe wave conditions in the Gulf of Mexico; and,

- the more sophisticated fire detection and fire-fighting systems of the VULCANUS as well as the generally less flammable nature of the cargo relative to the cargos ordinarily carried by tank ships; e.g., crude oil, gasoline, naptha or jet fuel.

Thus, the non-impact spill rates used herein should be viewed as reasonable "upper-bound" rather than "best" estimates of the non-impact spill rates for the VULCANUS.

6. Summary of Spill Rates. Table 6 provides a summary of expected spill rates by type of casualty and location. Non-impact casualties are estimated to represent approximately 48 percent of all casualties. For reasons discussed in Section III.5 this overstates the relative frequency of spill events emanating from this type of casualty. Collisions represent about 28 percent of the total while groundings represent approximately 17 percent of the total. Rammings account for the remaining 6 to 7 percent.

On the basis of location, the pier and harbor location represents about 38 percent; the Mobile Bay location represents approximately 12 percent; the coastal location represents approximately 31 percent; and, the burn zone location represents the remaining 19 percent.

IV. DISTRIBUTION OF SPILL SIZES

One method of estimating the size distribution of spills is to use an engineering approach, such as the Minorsky method.[16] This method provides an estimate of the three dimensional (longitudinal, transverse, and vertical) extents of damage to ship when struck at some particular location along the ship's length. For any particular location, the extent of damage (and thus, the breaching of any cargo tank or combination thereof) is estimated as a function of ship mass, velocity at impact, and angle of impact and the resistance of the ship's structure within that area. When such extents of damage are integrated with the distribution of the locations of the centers of damage along the ship, it is theoretically possible to develop the probabilistic distribution of cargo involvement given a particular type of impact casualty. When this is further integrated with the size of the cargo tanks involved, a size distribution of expected spills may be developed. As might be imagined, the computational effort required to perform these calculations is extensive.

Another approach is to use historical tank ship spill data to directly develop probabilistic spill size distributions given the occurrence of a spill. However, such spill data is overwhelmingly skewed towards the conventional, single hull type oil tanker with generally no less than five centerline tanks and five pairs of wing tanks and an average, single cargo tank capacity which approximates the total cargo capacity of the incineration ship. Thus, to estimate spill volume distributions for the incineration ship from such data would be extremely difficult.

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[16] Minorsky, V.O., "An Analysis of Ship Collisions with Reference to Protection of Nuclear Power Plants," Journal of Ship Research, October, 1969.

We think it is more reasonable to estimate spill volume distributions for the incineration ship by using an analytical approach that considers three combinations of cargo tank involvement following a casualty; i.e., a single cargo tank involvement; a two, adjacent cargo tank involvement; and, a three or more cargo tank (including the loss of the entire cargo). In the case of collision, grounding, and ramming spill events, it is most probable (on the order of 80 percent) that given the occurrence of one of these events, only one cargo tank will be affected. For a smaller percentage of these spill event types (approximately 15 percent) , it is probable that due to the extent and location of the damage which crosses either a longitudinal or transverse bulkhead between cargo tanks, two adjacent cargo tanks will be affected. Since the incineration ship is designed to remain afloat with the loss of two adjacent compartments (i.e., the "two compartment standard" of subdivision and damage stability throughout the cargo length), accidents which inflict damage to either one or two tanks would not cause the ship to sink. Lastly, for some (approximately 5 percent) impact casualties, it is conceivable that three or more (including the totality of) cargo tanks may be affected and its contents released to the sea.

For the purposes of this analysis, the average cargo tank volume of the incineration ship is taken at 400 cubic meters (m^3). Therefore, for impact type casualty events, in the case of a one cargo tank involvement, the limiting spill volume is 400 m^3; in the case of a two cargo tank involvement, the limiting spill volume is 800 m^3; and, in the case of a three or more cargo tank involvement, the limiting spill volume is 3,170 m^3 or the total volumetric capacity of the cargo tanks. As a matter of note, in the cases of grounding or ramming events, it is less likely that the entire contents of affected cargo tanks will be released since the cargo may not be in free communication with the sea as would ordinarily be the case with collision damage. However, in the interest of conservatism, the entire contents of a cargo tank is considered to be released from any affected cargo tank regardless of impact casualty type.

There is little information from which to estimate the size distribution of spills from non-impact casualties. Accordingly, it is assumed that the size distribution would be the same as for impact casualties, but there is obviously some amount of uncertainty regarding this assumption.

V. SUMMARY

As previously discussed in Part III of this Appendix, Table 6 provides a summary of the expected spill rates by type of casualty and by location for the incineration ship operation in Mobile. Overall, the expected spill rate has been estimated to be approximately six in 100,000 voyages or about one spill event per 16,600 voyages. Given the occurrence of a spill event, it is most likely (i.e., a conditional probability of 0.80) that the spill volume will be equal to or less than (≤) the contents of a single cargo tank or ≤ 400 m^3. There is a conditional probability of 0.15 that the spill volume will be ≤ 800 m^3 from a two tank involvement and a conditional probability of 0.05 that the spill volume will be ≤ 3,170 m^3 (the total tank capacity) from a three or more tank involvement.

To place the expected spill rate value of six per 100,000 voyages in perspective, Table 7 presents spill rates by location and by type of casualty for worldwide tank ships of 2,000 deadweight tons and greater. That spill rate is approximately 6.1 spill events in 10,000 voyages or about one per 1,650 voyages. In other words, it appears that the incineration ship operation considered here would have an expected spill rate of about an order of magnitude less than tank ship operations throughout the world.

The distribution of expected spills for the incineration ship by location and by type of casualty is given in Table 8. From that table (which has been derived from Table 6) it can be seen that four "cells" of the sixteen cell matrix (4 types of casualties by four locations) constitute 60 percent of the total. Spills from non-impact casualties in the pier and harbor location account for 28 percent of the total; spills from collisions and groundings in the coastal location account for 21 percent of the total; and, spills from non-impact casualties in the burn zone location account for the remaining 11 percent of the total.

TABLE 1: INCINERATOR SHIP OPERATIONS

VULCANUS I

- Completed 145 incineration voyages between 1972 and 1983
- Incinerated 400,000 metric tons of waste
- Sailed 116,430 n. miles in ballast and 47,980 n. miles loaded
- 3,600-ton carrying capacity
- 2 incinerators with 15 to 18 ton/hour capacity

VULCANUS II

- Completed 27 incineration voyages since November, 1982
- Incinerated 75,000 metric tons of waste
- Sailed 6,920 n. miles in ballast and 7,600 n. miles loaded
- 3,600-ton carrying capacity
- 3 incinerators with 25 ton/hour capacity

VESTA

- Completed 150 incineration voyages since 1979
- Incinerated 180,000 metric tons of waste
- Sailed 30,000 n. miles in ballast and 33,000 miles loaded
- 1,200- to 1,300-ton carrying capacity
- 1 incinerator with 8 to 10 ton/hour capacity

MATHIAS I, II, III

- No information available

SOURCE: PRIVATE COMMUNICATION WITH B. DeBOER, CHEMICAL WASTE M'G'M'T, AUGUST 29, 1984

TABLE 2: EXPECTED SPILL RATES FOR COLLISIONS

	PIER and HARBOR	MOBILE BAY	COASTAL	BURN ZONE	TOTAL
HISTORICAL SPILL RATE (PER VOYAGE) [a]	1.79E-05	2.50E-05	7.14E-06	5.00E-06	5.50E-05
DOUBLE HULL	0.80	0.91	0.95	0.95	---
CP PROPELLER/ BOW THRUSTER	0.75	0.89	0.94	0.94	---
USCG NAVIGATION REQUIREMENTS	0.40	0.10	1.00	1.00	---
EXPECTED SPILL RATE (PER VOYAGE)	4.30E-06	1.92E-06	6.38E-06	4.47E-06	1.71E-05

[a] SOURCE: ECOTANK, ECO, INC., ANNAPOLIS, MD., (2000 TO 10000 DEADWEIGHT)

TABLE 3: EXPECTED SPILL RATES FOR GROUNDINGS

	PIER and HARBOR	MOBILE BAY	COASTAL	BURN ZONE	TOTAL
HISTORICAL SPILL RATE (PER VOYAGE) [a]	1.61E-05	2.68E-05	3.21E-05	0.00E+00	7.50E-05
U.S. GULF COAST	0.31	0.31	0.31	---	---
DOUBLE HULL	0.10	0.60	0.78	---	---
CP PROPELLER/ BOW THRUSTER	0.50	0.75	0.86	---	---
USCG NAVIGATION REQUIREMENTS	0.90	0.90	1.00	---	---
EXPECTED SPILL RATE (PER VOYAGE)	2.25E-07	3.36E-06	6.68E-06	0.00E+00	1.03E-05

[a] SOURCE: ECOTANK, ECO, INC., ANNAPOLIS, MD., (2000 TO 10000 DEADWEIGHT)

TABLE 4: EXPECTED SPILL RATES FOR RAMMINGS

	PIER and HARBOR	MOBILE BAY	COASTAL	BURN ZONE	TOTAL
HISTORICAL SPILL RATE (PER VOYAGE) [a]	3.57E-06	3.57E-06	3.57E-06	0.00E+00	1.07E-05
DOUBLE HULL	0.10	0.60	0.78	---	---
CP PROPELLER/ BOW THRUSTER	0.50	0.75	0.86	---	---
USCG NAVIGATION REQUIREMENTS	0.90	0.90	1.00	---	---
EXPECTED SPILL RATE (PER VOYAGE)	1.61E-07	1.45E-06	2.39E-06	0.00E+00	4.00E-06

[a] SOURCE: ECOTANK, ECO, INC., ANNAPOLIS, MD., (2000 TO 10000 DEADWEIGHT)

TABLE 5: EXPECTED SPILL RATES FOR NON-IMPACT TYPE CASUALTIES

	PIER and HARBOR *	MOBILE BAY **	COASTAL	BURN ZONE	TOTAL
HISTORICAL SPILL RATE (PER VOYAGE) [a]	1.79E-05	3.57E-07	3.57E-06	7.14E-06	2.89E-05
EXPECTED SPILL RATE (PER VOYAGE)	1.79E-05	3.57E-07	3.57E-06	7.14E-06	2.89E-05

* Most of the events are likely to occur at the pier as opposed to the harbor.

** Non-pact type casualties are a function of time rather than traffic density or proximity to land for impact type casualties. Since the ship time in Mobile Bay is ten percent of the estimated time in the coastal area, the expected non-impact spill rate is one-tenth the rate of the coastal areas.

[a] SOURCE: ECOTANK, ECO, INC., ANNAPOLIS, MD., (2000 TO 10000 DEADWEIGHT)

TABLE 6: EXPECTED SPILL RATES BY LOCATION BY TYPE OF CASUALTY

	PIER and HARBOR	MOBILE BAY	COASTAL	BURN ZONE	TOTAL
COLLISIONS	4.30E-06	1.92E-06	6.38E-06	4.47E-06	1.71E-05
GROUNDINGS	2.25E-07	3.36E-06	6.68E-06	0.00E+00	1.03E-05
RAMMINGS	1.61E-07	1.45E-06	2.39E-06	0.00E+00	4.00E-06
NON-IMPACT TYPE CASUALTIES	1.79E-05	3.57E-07	3.57E-06	7.14E-06	2.89E-05
TOTAL EXPECTED SPILL RATE (PER VOYAGE)	2.26E-05	7.09E-06	1.90E-05	1.16E-05	6.03E-05

TABLE 7: SPILL RATES FOR TANK SHIPS BY LOCATION BY TYPE OF CASUALTY

	PIER and HARBOR	MOBILE BAY	COASTAL	BURN ZONE	TOTAL
COLLISIONS	3.75E-05	6.62E-05	7.76E-05	5.20E-06	1.87E-04
GROUNDINGS	3.32E-05	6.85E-05	6.88E-05	0.00E+00	1.71E-04
RAMMINGS	6.01E-05	1.36E-05	3.53E-06	1.30E-06	7.85E-05
NON-IMPACT TYPE CASUALTIES	2.37E-05	2.60E-06	2.65E-05	1.24E-04	1.77E-04
TOTAL SPILL RATE (PER VOYAGE)	1.55E-04	1.51E-04	1.76E-04	1.31E-04	6.13E-04

SOURCE: ECOTANK, ECO, INC., ANNAPOLIS, MD.

TABLE 8: DISTRIBUTION OF EXPECTED SPILLS BY LOCATION BY TYPE OF CASUALTY

	PIER and HARBOR	MOBILE BAY	COASTAL	BURN ZONE	TOTAL
COLLISIONS	0.071	0.032	0.106	0.074	0.284
GROUNDINGS	0.004	0.056	0.111	0.000	0.171
RAMMINGS	0.003	0.024	0.040	0.000	0.066
NON-IMPACT TYPE CASUALTIES	0.297	0.006	0.059	0.118	0.479
ALL CASUALTY TYPES	0.375	0.118	0.315	0.192	1.000

NOTE: DERIVED FROM TABLE 6

Appendix D

ESTIMATION OF RISKS RESULTING FROM INHALATION AND INGESTION OF HAZARDOUS EMISSIONS FROM LAND-BASED INCINERATORS

This appendix describes the estimation of human health risks from exposure to PCB and EDC emissions around hypothetical incinerators located in Arkansas and Texas. Health risks are calculated for the inhalation route of exposure and for exposures resulting from the ingestion of foodstuffs contaminated with pollutants emitted from the incinerators. Risks are estimated for exposure to principal organic hazardous constituents (POHCs), products of incomplete combustion (PICs) and metals. Both stack and fugitive emissions are considered.

Our description of the risk estimation procedure used in this analysis is divided into six sections:

1. Methodology and assumptions used to model human exposures to incinerator emissions,
2. Results from the exposure modeling and a comparison of these results with those from analyses at 32 other hazardous waste incinerators in the U.S.,
3. Methodology and assumptions used to convert the inhalation exposure to human health risks,
4. Analysis of inhalation risk estimates,
5. Methodology for assessing ingestion risks,
6. Analysis of ingestion risk estimates, and
7. Assessment of comparable risks.

The risk assessment approach described in this appendix relies on a variety of analytic techniques and computer models. Wherever possible we have used practices developed or accepted by EPA. These analytic techniques and models are not, however, perfect predictors of human health risks.

When using the results of our analysis, one should be aware of the major assumptions and limitations underlying the approach.

First, the risk assessment only considers health risks to persons living within 50 kilometers of the incinerator. For estimates beyond this distance, generally accepted modeling techniques are not available. Pollutants which are transported beyond the 50 kilometer boundary are removed from further consideration in the analysis. For persistent pollutants, this may cause an underestimate of risks. Second, field verification tests at the sites selected for our analyses have not been performed for the exposure estimation models. Calibrating the models for these sites would produce more accurate estimates of exposures. Third, the emission estimates for PICs and POHCs are based not on actual measurements at specific facilities, but on regulatory requirements (POHCs) and trial burn data from a sample of facilities (PICs and metals). Actual measurements of PIC, POHC, and metal emissions for the wastes of interest would also provide more accurate estimates of exposures. Fourth, for the ingestion analyses the modeling techniques used to estimate exposure are extremely complicated resulting in greater uncertainty in these estimates than in the estimates of inhalation risks.

HUMAN EXPOSURE ANALYSIS: ASSUMPTIONS AND METHODOLOGY

Location Assumptions

As noted above, air dispersion analyses were performed for hypothetical incinerators located in Texas and Arkansas. These sites were selected in part because they are representative of the meteorology at locations where PCB incineration is currently permitted. In addition, selection of these sites allows us to estimate impacts of incineration for both high and low population density regions. In 1980 the number of persons living within 50 kilometers of the Texas and Arkansas sites was 2,500,000 and 91,000 respectively.

Waste Feed and Emission Rate Assumptions

Air dispersion analyses were performed separately for PCB and EDC wastes. For the PCB waste we assumed a waste feed rate of 56,000 metric tons per year.1/ Thirty-five percent (19,600 MT/yr) of this waste was assumed to be PCBs. We assumed the remaining volume to be non-hazardous. A destruction and removal efficiency (DRE) of 99.9999 percent was used to calculate the POHC emission rate for incinerators burning the waste. This resulted in annual PCB stack emissions of .0196 metric tons. For the EDC waste, we assumed a feed rate of 68,400 metric tons per year of which fifty percent was EDC (34,200 MT/yr). We assumed the remaining volume to be non-hazardous. For incinerators burning the EDC waste, a DRE of 99.99 percent is used to calculate annual stack emissions of EDC (3.42 MT/yr).

We also modeled PIC emissions for incinerators at the two sites. The PIC emission rates used in this analysis are average rates observed in trial burns of PCB and non-PCB organic wastes. (For a more detailed description of the calculation of PIC emission rates, see Appendix E.) The PIC emission rate used in the dispersion analyses for PCB wastes is 3.40E-8 percent of the waste feed rate. For the EDC waste PIC emissions were fixed at 8.94E-4 percent of the feed rate.

In our analysis of risks from incinerator stack emissions, we also modeled metal emissions. Given the metals assumed present in the waste streamc, we focused the analysis on four metals which EPA considers to be carcinogenic; arsenic, cadmium, chromium, and nickel. This was done based on an initial screening anlysis which suggested that none of the non-carcinogenic metals potentially present in the waste would exceed its Acceptable Daily Intake, a measure of the dose level (mg/day) below which no adverse health effects are observed.2/

To determine emission rates for metals, we reviewed data on trial burns conducted at land-based incinerators. (Performance Evaluation of Full-Scale Hazardous Waste Incinerators, Midwest Research Institute, Kansas City, Missouri, 1984.) These data indicate a wide range of emission rates for three of the carcinogenic metals. At facilities with some form of air pollution control equipment, the following range of emissions was found:

1/Although no commercial incinerators currently permitted to burn PCBs are this large, facilities of this size are feasible.

2/This analysis was based on Acceptable Daily Intakes published by EPA's Environmental Criteria and Assessment Office, September 30, 1982.

	Range (% of waste feed)
Cadmium	1 - 50
Chromium	< 1 - 34
Nickel	< 1 - 36

These emissions represent only those metals emitted as particulates. Actual emissions could be higher if any of these compounds were also emitted in gaseous form.

Given the wide range of potential emissions of these three metals, selection of an appropriate emission factor was difficult. Instead of using a worst or best case value, we selected an emission rate of 10 percent of the waste feed for each of the three metals. This falls between the worst case (50 percent) and the best case (< 1 percent). Because metal emission rates do not cluster neatly around an average value, readers should recognize that there is significant uncertainty associated with the 10 percent emission rate for metals.

Estimates of arsenic emissions were not available in the trial burn data. Arsenic is a relatively volatile metal and is likely to be emitted from incinerators as a gas. Typical air pollution control equipment at hazardous waste incinerators may not be effective in removing arsenic fumes from the stack gases. Based on discussions with Dr. Ben Blaney of the Incineration Research Branch of EPA's Office of Research and Development, we concluded that an arsenic emission rate of 50 percent of the waste feed quantity of the metal would serve as a reasonable approximation given the paucity of actual data. In a worst case scenario where 100 percent of the arsenic is emitted from the stack, risks would increase by a factor of two.3/

Fugitive emissions from the incinerator storage and transfer operations were estimated by Arthur D. Little, Inc. (see Appendix B). For the dispersion analysis, fugitive emissions were assumed to be 0.197 metric tons per year for the PCB waste and 0.292 metric tons per year for the EDC waste.

3/Readers who believe the 50 percent arsenic emission rate to be inappropriate can linearly scale the arsenic risk estimates provided in Exhibit D-9 to reflect other emission rate assumptions.

Air Dispersion Modeling Assumptions

To estimate pollutant dispersion from the Texas and Arkansas incinerators, we used the ATM atmospheric dispersion model (see The Atmospheric Transport Model for Toxic Substances (Draft), R.S. Raridon, B.D. Murphy, W.M. Culkowski, and M.R. Patterson, Computer Sciences Division, Oak Ridge National Laboratory). ATM is a gaussian plume climatological model which is linked to 1982 U.S. census data and Stability Tabular Array (STAR) data from 394 weather stations in the continental United States. The model estimates annual average pollutant concentrations at 160 receptor points located on a polar coordinate system. The receptors are arrayed along 16 compass points at distances of .5, 1, 2, 3, 4, 5, 10, 15, 25, and 50 kilometers from the source. The model also estimates the population at each receptor point. Since each receptor actually represents a geographic area and not just a point, the total number of people living in that area are assigned to that point.

As input the ATM model requires the following information:

- Source location (latitude/longitude),
- Source height,
- Source radius,
- Exit gas velocity,
- Exit gas temperature,
- Mass emission rate,
- Pollutant state (gas/particulate),
- Dispersion coefficient estimation technique, and
- Weather station.

Separate runs of the ATM model were performed for stack gas, fugitive, and metals emissions. Wet and dry deposition was modeled for all gases (PCB, EDC, PICs) and particulates (arsenic, cadmium, chromium, and nickel).

For each of the model input parameters, the values used in the stack gas dispersion analyses are listed in Exhibit D-1. The stack height and diameter and the source exit gas temperature and velocity are representative of currently permitted PCB incinerators. The mass emission rate (grams/second) was calculated by dividing the annual emission rate (grams) by the number of seconds in a year (3.15E+7). PCB, EDC and PIC emissions are assumed to be gaseous. We assume metals are emitted as particulates. To model particulates, the ATM model requires an estimate of the particulate size and density. No data were available to predict the actual size of metal particulates emitted from a hazardous waste incinerator. In the absence of incinerator data, we used data on the size of particulates found in measurements of atmospheric background concentrations of metals. As an input to the ATM, we used the median particulate diameter averaged over the four metals. The density of this particulate was assumed to be the average density of arsenic, cadmium, chromium, and nickel. In modeling both sites, STAR data for the weather station closest to the source were used.

For fugitive emissions, a different set of source and emission parameters was used (Exhibit D-2). Pollutant concentrations are sensitive to the size of the area from which the fugitives are emitted. Since various facility configurations could exist, we modeled the fugitives as if they were emitted from a point source (a worst-case scenario based on the assumption that all the fugitives would be emitted from a small area). Fugitive emissions were assumed to be non-buoyant. Thus they are emitted at a velocity close to zero (.001 m/s) and a temperature near ambient levels (293 degrees Kelvin). A source radius of 0.56 meters was used. Other input parameters were the same as for the stack gas runs of the model.

EXPOSURE MODELING RESULTS

The results from the dispersion modeling analysis for the Texas and Arkansas incinerators are presented for PCB and EDC wastes in Exhibits D-3 and D-4 respectively. To illustrate the relative impact of different types of emissions, we provide separate estimates for stack emissions of PICs, POHCs, and metals and for fugitive emissions.

In these exhibits three measures of dispersion and exposure are presented. Aggregate exposure is the product of ambient air concentration and population summed over all the receptor points. This metric is a predictor of total human exposure to the pollutants in the region of concern. The second measure is the ambient air concentration of the pollutant at the receptor location with the highest overall pollutant risk. A person living at the maximum concentration location faces the highest risks from incinerator emissions. However, since the maximum often occurs at a receptor point where no one actually lives, we also report a third measure of exposure, the most exposed individual (MEI) concentration. MEI concentration is the highest ambient air concentration of the pollutant at a currently populated receptor point.

The dispersion analysis results suggest that exposures, both aggregate and maximum, are higher at the Texas site than at the Arkansas site. The higher aggregate exposures are explained by the fact that the Texas site has a total population approximately 28 times greater than Arkansas'. The higher maximum concentration results from the lower stack assumed for the Texas facility. For the MEI, exposures are higher at the Arkansas site than in Texas. This results from people living closer to the facility than they do in Texas. The data also indicate that at both sites EDC exposures are higher than those for PCBs. The reason for this is the higher DRE (99.9999 versus 99.99) required for PCB wastes and the higher EDC throughput (34,200 MT/yr versus 19,600 MT/yr). Due to their low removal efficiencies, metals emissions account for most of the exposure at both the Texas and Arkansas sites.

The relative magnitude of these results can be seen by comparing them with the aggregate exposure and MEI concentrations predicted in dispersion analyses reported by the Midwest Research Institute (MRI) for 32 incinerators operating in the United States (see _Inhalation Exposure Methodology Modeling of Hazardous Waste Incinerators_, MRI, 1983). These analyses considered only POHC emissions from incinerator stacks. The results from the MRI study are compared in Exhibit D-5 with those from our Texas and Arkansas analyses. Aggregate exposures for the MRI case studies range from 1.32E-1 to 1.24E+4 with a median value of 2.14E+2. Comparing these with the aggregate exposures due to POHC emissions from stacks at the Texas and Arkansas sites, our analysis suggests that, except for EDC exposures at the Texas site, exposures are below the median for the MRI case studies. While the Texas EDC exposure is above the case study median, it is still only 5 percent of the maximum value observed in the MRI study.

Exhibit D-5 also permits comparison of MEI exposures at the MRI trial burn sites with those in Texas and Arkansas. For the PCB and EDC wastes, the MEI concentrations at both sites are below the median value for the case studies. This suggests that, with respect to emissions of stack POHCs, incineration at the Texas and Arkansas sites used in our analyses is somewhat less risky for the most exposed individuals than at other hazardous waste incinerators in the U.S.

RISK ESTIMATION METHODOLOGY

The methodology used to estimate human health risks from inhalation of hazardous air pollutants is illustrated in Exhibit D-6. To derive estimates of cancer incidence assuming 70-year exposure the aggregate dose (mg/kg/day) is estimated by multiplying aggregate exposure (person-ug/m3) from the air dispersion analysis by a conversion factor which takes into consideration the daily volume of air a person breathes (22.8 m3), a person's average weight (70 kg) and the proportion of the pollutant which is absorbed into the body (65 percent) (personal communication, Dr. Michael Dourson, EPA Environmental Criteria and Assessment Office, September 1984). The aggregate dose is then multiplied by the chemical's cancer unit risk factor, a measure of risk generated by a dose of 1 milligram per kilogram of bodyweight per day over 70 years. Average lifetime risk is estimated by dividing the incidence by the exposed population.

The calculation of risk to the most exposed individual or risk at the maximum concentration location is also illustrated in Exhibit D-6. For this calculation, the ambient air concentration of the pollutant is converted into a dose (mg/kg/day) using the same assumptions as were used in the estimate of incidence. The individual dose is then multiplied by the unit risk factor to estimate individual lifetime risk.

In our analysis, incidence and risk estimates are calculated for exposures to PICs, POHCs, and metals in stack gases and to fugitive emissions. All cancer unit risk factors are estimates of EPA's Cancer Assessment Group. The unit risk factors used for the various types of emissions are presented in Exhibit D-7. For POHC and fugitive emissions, we used the unit risks for the primary pollutant (either PCB or EDC). For metal emissions from the incinerator stack, we evaluated risks using the CAG factors for arsenic, cadmium, chromium (hexavalent), and nickel. For PICs, the determination of the appropriate risk factor was complicated by the large number of compounds that appear as PICs in stack gases. For the PCB wastes, the PIC's identified in the trial burns were dioxins and furans. In estimating the risk from exposures to mixtures of dioxins and furans, we followed EPA recommendations by applying the unit risk factor for 2,3,7,8-TCDD to the dose of TCDDs (personal communication, Dr. Donald Barnes, EPA Office of Toxic Substances, September 1984).

In the trial burns on which we based our estimates of PIC emission rates for non-PCB organics, 35 chemicals were detected. Seven of these compounds are carcinogens with unit risk factors ranging from 1.20E-2 to 1.67E0. Sixteen of the 35 chemicals were non-carcinogens with acceptable daily intakes ranging from .014 to 880 milligrams per day. Health effects data were not available for the remaining compounds.

Chloroform was detected in 5 of the 8 trial burns analyzed, and on average chloroform accounted for 60 percent of all PIC emissions in these trial burns. Based on the large percentage of chloroform in the PIC emissions, we used its unit risk factor in our estimates of risks from PICs in the EDC analyses. The unit risk for chloroform is approximately the median value for the carcinogens appearing in the trial burn analyses of PICs.

ANALYSIS OF INHALATION PATHWAY RISK ESTIMATES

Exhibit D-8 summarizes the human health risks due to inhalation of hazardous pollutants emitted by land-based incinerators. Four measures of health risk are calculated. Incidence is an estimate of the total number of cancer cases which would occur if people living in the study area were exposed for a period of 70 years to the average annual pollutant concentrations estimated in our dispersion analyses. Incidence serves as a measure of the total health impact of the incinerator emissions. The total incidence of cancer at the two sites ranges from 0.29 to 0.43 cases over 70 years of continuous incinerator emissions. In general, the total number of cases at the Texas and Arkansas sites is remarkably similiar. In both cases, metal emissions account for most of the risk. When PCBs are burned at the Texas site, metals are responsible for 70 percent of the risk. For the other waste/site combinations, metals account for more than 91 percent of the risk. The contributions of individual metals are shown in Exhibit D-9.

For the PCB wastes, PIC emissions are the second most important contributor to risk. For the EDC wastes, POHCs contribute slightly more than PICs to risks. Fugitive emissions are not a major factor for any of the waste/site combinations.

The second health metric in Exhibit D-8 is average risk which represents the average probability that a person living in the study area would develop cancer after 70 years of exposure to the incinerator emissions. Average risk is calculated by dividing the incidence (cases) by the population in the study area. Average risks at the two sites ranged from 1.50E-7 to 3.60E-6. At the upper end of this range, a person living near the Arkansas incinerator has roughly a 4 in 1,000,000 chance of developing cancer after 70 years of exposure.

The third measure of human health risk provided in Exhibit D-8 is the maximum individual risk. This is a measure of the risk at the currently populated receptor point with the highest ambient air concentration of the pollutant. It is a rough measure of the highest risk which would be faced by anyone in the study area given the current population distribution. Our estimates of maximum individual risks range from 2.37E-5 to 3.65E-5. These are only approximate indicators of MEI risks because the technique for allocating population to receptor points itself only approximates where people currently live.

Because of the approximate nature of the MEI estimates, we report a fourth measure of health risk, the maximum concentration risk. This is the risk at the receptor point with the highest overall health risk. In our analyses, no persons were reported to be living at the maximum concentration locations. Maximum concentration risks at the Texas and Arkansas sites ranged from 4.11E-5 to 2.65E-4. At the upper end of the range, the maximum concentration risk is approximately ten times the MEI risk.

Overall, Exhibit D-8 illustrates that metals account for most of the inhalation risk associated with land-based incineration MEI. For the PCB wastes, metals account for 90 percent of the MEI risk at the Texas facility and 92 percent of the risk at the Arkansas location. For the EDC wastes, metals contribute in excess of 99 percent of the MEI risk at both sites. These same patterns also apply to the maximum concentration risks.

In general, the risks to the most exposed individual are roughly comparable at the two sites regardless of the waste. However, the maximum concentration risk is approximately one order of magnitude greater at the Texas site. The higher maximum concentration risk in Texas is a result of the shorter stack assumed at the Texas facility. This causes higher maximum concentration than at the Arkansas facility. This does not, however, result in a higher MEI risk in Texas because the MEI receptor point is farther from the incinerator than it is in Arkansas.

METHODOLOGY FOR ASSESSING INGESTION RISKS

Ingestion of contaminated foodstuffs is a second possible route by which people may be exposed to incinerator emissions. Our analysis of the relative risks of the inhalation and ingestion routes of exposure is based on two recent studies of foodchain exposures resulting from hazardous waste incinerator emissions (see Multiple Pathways Screening Level Assessment of a Hazardous Waste Incineration Facility (Draft), G.A. Holton, C.C. Travis, E.L. Etnier, F.R. O'Donnell, D.M. Hetrick, and E. Dixon, Oak Ridge National Laboratory, September 1984, and A Comparison of PCB Exposures to Human Populations from Oceanic and Terrestrial Incineration, G.A. Holton, C.C. Travis, E.L. Etnier, Life Systems, Inc., September 1984).

These studies estimate the relative doses attributable to the ingestion and inhalation exposure pathways at three agriculturally productive sites in the U.S. For the EDC ingestion analysis, exposures were estimated for three sites; 1) Southern California, 2) Upper Midwest, and 3) Central Midwest. For the PCB analysis, exposures are only calculated for the Central Midwest site. In both studies, effects were modeled out to 100 kilometers. Inhalation dose estimates are calculated using standard dispersion modeling techniques and human uptake assumptions. Although different stack parameters, population, and meteorologies were used, the inhalation dose methodology is similar to the one used in analyses we performed for the Texas and Arkansas sites.

For the ingestion analysis in the two studies, a two step process was used to estimate exposures. First, based on pollutant deposition rates, uptake rates for agricultural crops, and bioconcentration factors for animals consuming these crops, pollutant concentrations in milk, beef, and seven other grain and produce crops were calculated. Then, using U.S. Department of Agriculture (USDA) dietary assumptions, the researchers estimated human doses of the contaminants from ingestion of these foods. It was assumed that persons living in the study area would consume food grown in the area. In cases where not enough of a specific dietary component was produced locally, the researchers substituted non-contaminated food imported from regions outside the study area.

In Exhibit D-10, we summarize the dose estimates from the two studies for the EDC and PCB wastes. The results suggest that the relative doses attributable to inhalation and ingestion are compound specific. For the PCB wastes, ingestion represents 6.5 percent of the total dose whereas for EDC it is only 4.7 percent.

To estimate ingestion risks at the Texas and Arkansas sites, we assumed that the ratios of ingestion to inhalation doses are equal to the average values in Exhibit D-10. We then scaled the air doses resulting from stack emissions of POHCs at the two sites by the inhalation to ingestion dose ratio to obtain Texas and Arkansas ingestion doses. The ingestion analysis is based on the POHC emissions because data on the effects of metals and PICs were not assessed in the two studies of foodchain exposures. These doses were multiplied by the unit risk factors for ingestion of EDC and PCB (see Exhibit D-7) to determine cancer incidence from foodchain exposures. Average risk can be calculated by dividing the incidence by the population in the study area. Maximum and MEI foodchain risks could not be calculated from available data.

ANALYSIS OF INGESTION PATHWAY RISK ESTIMATES

Exhibit D-11 summarizes the relative risks due to ingestion and inhalation at our case study sites. The cancer incidence resulting from ingestion is relatively insignificant ranging from 3.83E-5 to 5.03E-4 cases over 70 years. Average risks from ingestion are also very low, ranging from 9.15E-11 to 9.17E-10. In terms of contribution to total incidence or average risk, ingestion of the EDC-contaminated food is approximately three times more important than ingestion of PCB-containing foods. For EDC incineration at Texas and Arkansas sites, ingestion accounts for between 0.03 and 0.13 percent of the total risk, whereas ingestion of PCBs accounts for only 0.01 to 0.05 percent of the total risk. Although PCBs have a high bioconcentration factor, foodchain exposures to them are relatively low due to the tendency of PCB compounds to bind tightly to organic materials in the soil. Overall, the majority of the risk is attributable to inhalation regardless of the waste or site.

The following limitations apply to this ingestion risk analysis. First, the assumptions underlying the ingestion/inhalation dose ratios are not necessarily appropriate for the Texas and Arkansas sites. The Texas and Arkansas ingestion risk estimates are based on dose ratios calculated for different regions of the country. Food production around our two sites may differ significantly from that in the highly productive regions on which the dose ratios are based. As a result, our ingestion estimates may represent upper bounds on the ingestion risks at the two sites.

In addition, the stack and population parameters on which the ingestion to inhalation dose ratios are based are different from those used in our analyses. Further, the PCB isomer (Arochlor 1242) used to estimate ingestion to inhalation dose ratios is different from the PCB isomer (Arochlor 1254) assumed in our analysis. In addition, we assume 100 percent absorption of the compound whereas the other studies used 95 percent. It is unclear whether these inconsistencies result in over- or underestimates of risk for the Texas and Arkansas sites. Finally, effects were modeled out to 100 kilometers in the ingestion analyses on which we base our estimates. However, because we only use the ratio of ingestion to inhalation doses, not the absolute values, our results assuming a 50 kilometer boundary should be reasonable estimates of ingestion risks.

All of our ingestion doses only consider POHC stack emissions. In the studies on which we base our analysis, researchers assumed that fugitive emissions did not significantly contribute to ingestion risk. They also did not consider ingestion of PICs or metals. Because of the compound specific nature of the ingestion estimates, we were not able to scale the ingestion results to include concentrations of PICs and metals in foodstuffs. This may lead to an underestimate of the total ingestion dose.

ASSESSMENT OF COMPARABLE RISKS

To place the human health risks from land-based incineration of PCB and EDC wastes in perspective, we compare these risks with those resulting from exposure to other hazardous air pollutants. A recently published EPA report (The Magnitude of the Air Toxics Problem in the United States (Draft), E. Haemisegger, A. Jones, B. Steigerwald, V. Thomson, September 1984) summarizes estimates available on the risks from exposure to airborne toxics.

These risk estimates are presented in Exhibit D-12 for the 19 pollutants or pollutant groups which cause significant risks. Risk estimates are based primarily on ambient air concentrations reported in three recent EPA studies. Two of the studies (the

NESHAPS and 35 County studies) are based on air dispersion modeling analyses and the third reports air pollutant concentrations from ambient air quality monitoring at approximately 170 sites in the United States. These estimates suggest that on average, persons in the U.S. face lifetime cancer risks of between 3.9E-4 and 6.1E-4 as a result of inhalation of these 19 hazardous air pollutants.

In Exhibit D-13 we compare these risk estimates with those from our analyses of maximum individual risks at the Texas and Arkansas sites. Our estimates of maximum risks at the two sites suggest that relative to the risks reported in the air toxics study above, the incremental risk from land-based incineration of the EDC and PCB wastes at the Texas and Arkansas sites is small. In the worst case, if we calculate the incremental contribution to the MEI risk from burning the EDC waste at the Arkansas site with the average air toxics risk estimate from the 35 County Study, we conclude that total risks increase less than by 10 percent. For the other three cases the percentage increases are even smaller. Note that this 10 percent increase in risk applies to the most exposed individual. Other persons would receive an even smaller increase in their average risk from air toxics.

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Exhibit D-1

ASSUMPTIONS FOR AIR MODELING ANALYSIS
STACK EMISSIONS

	Texas	Arkansas
Source Location		
Latitude	N 29 43 35	N 33 12 19
Longitude	W 95 05 40	W 92 37 45
Weather Data Source	Ellington/ Houston	El Dorado/ Goodwin
Waste Feed Rate (MT/yr)		
PCB	56,000	56,000
EDC	68,400	68,400
Percent DRE		
PCB	99.9999	99.9999
EDC	99.99	99.99
Source Height (m)	30.4	61.0
Source Radius (m)	0.90	1.45
Exit Gas Velocity (m/s)	7.2	4.6
Exit Gas Temperature (Kelvin)	323	336
POHC Emission Rate (g/s)		
PCB	6.22E-4	6.22E-4
EDC	1.08E-1	1.08E-1
PIC Emission Rate (g/s)		
PCB	6.04E-7	6.04E-7
EDC	1.94E-2	1.94E-2
Metals Emission Rate (g/s)		
PCB - Arsenic	8.88E-2	8.88E-2
Cadmium	1.77E-2	1.77E-2
Chromium	1.77E-2	1.77E-2
Nickel	1.77E-2	1.77E-2
EDC - Arsenic	1.08E-1	1.08E-1
Cadmium	2.17E-2	2.17E-2
Chromium	2.17E-2	2.17E-2
Nickel	2.17E-2	2.17E-2
Pollutant State		
POHCs and PICs	Gas	Gas
Metals	Particle	Particle
Particle Diameter (um)	1.9	1.9
Particle Density (g/cm3)	7.6	7.6
Dispersion Coefficient Estimation	Briggs	Briggs

Exhibit D-2

ASSUMPTIONS FOR AIR MODELING ANALYSIS
FUGITIVE EMISSIONS

	Texas	Arkansas
Source Location		
Latitude	N 29 43 35	N 33 12 19
Longitude	W 95 05 40	W 92 37 45
Weather Data Source	Ellington/ Houston	El Dorado/ Goodwin
Waste Feed Rate (MT/yr)		
PCB	56,000	56,000
EDC	68,400	68,400
Percent DRE		
PCB	99.9999	99.9999
EDC	99.99	99.99
Source Height (m)	2.0	2.0
Source Radius (m)	0.56	0.56
Exit Gas Velocity (m/s)	.001	.001
Exit Gas Temperature (Kelvin)	293	293
POHC Emission Rate (g/s)		
PCB	6.25E-3	6.25E-3
EDC	4.26E-3	9.26E-3
Pollutant State	Gas	Gas
Dispersion Coefficient Estimation	Pasquill Gifford	Pasquill Gifford

Exhibit D-3

EXPOSURE LEVELS FROM PCB WASTE

	Aggregate Exposure (person-ug/m3)		Maximum Concentration (ug/m3)		MEI Concentration (ug/m3)	
	Texas	Arkansas	Texas	Arkansas	Texas	Arkansas
Stack POHCs	3.62E0	5.99E-1	3.18E-4	5.79E-5	6.64E-5	4.53E-5
Stack PICs	3.51E-3	5.82E-4	3.09E-7	5.62E-8	6.46E-8	4.40E-8
Metals						
Arsenic	5.69E+1	5.04E+1	4.08E-2	7.11E-3	4.04E-3	5.62E-2
Cadmium	1.14E+1	1.01E+1	8.13E-3	1.42E-3	8.04E-4	1.12E-3
Chromium	1.14E+1	1.01E+1	8.13E-3	1.42E-3	8.04E-4	1.12E-3
Nickel	1.14E+1	1.01E+1	8.13E-3	1.42E-3	8.04E-4	1.12E-3
Fugitives	6.68E0	4.59E0	7.96E-3	1.69E-3	1.51E-4	1.38E-3
TOTAL	1.01E+2	8.59E+1	7.35E-2	1.31E-2	6.67E-3	1.04E-2

Source: IEc Analysis

Exhibit D-4

EXPOSURE LEVELS FROM EDC WASTE

	Aggregate Exposure (person-ug/m3)		Maximum Concentration (ug/m3)		MEI Concentration (ug/m3)	
	Texas	Arkansas	Texas	Arkansas	Texas	Arkansas
Stack POHCs	6.32E+2	1.05E+2	5.52E-2	1.01E-2	1.16E-2	7.91E-3
Stack PICs	1.13E+2	1.87E+1	9.92E-3	1.81E-3	2.08E-3	1.42E-3
Metals						
Arsenic	6.96E+1	6.15E+1	4.96E-2	8.64E-3	4.91E-3	6.84E-3
Cadmium	1.39E+1	1.23E+1	9.97E-3	1.74E-3	9.86E-4	1.37E-3
Chromium	1.39E+1	1.23E+1	9.97E-3	1.74E-3	9.86E-4	1.37E-3
Nickel	1.39E+1	1.23E+1	9.97E-3	1.74E-3	9.86E-4	1.37E-3
Fugitives	9.48E0	6.60E0	5.87E-3	1.19E-3	9.32E-5	9.72E-4
TOTAL	8.66E+2	2.29E+2	1.51E-1	2.70E-2	2.16E-2	2.12E-2

Source: IEc Analysis

Exhibit D-5

COMPARISON OF EXPOSURES: TEXAS, ARKANSAS, AND MRI CASE STUDIES

	Aggregate Exposure (person-ug/m3)	MEI Concentration (ug/m3)
PCB Waste		
Texas	3.62E0	6.64E-5
Arkansas	5.99E-1	4.53E-5
EDC Waste		
Texas	6.32E+2	1.16E-2
Arkansas	1.05E+2	7.91E-3
MRI Case Studies		
Median	2.14E+2	2.47E-2
Maximum	1.24E+4	3.54E-1
Minimum	1.32E-1	6.00E-4

Source: "Inhalation Exposure Methodology Modeling of Hazardous Waste Incinerators", Midwest Research Institute 1983.

Exhibit D-6

HEALTH RISK CALCULATIONS

A. Calculation of Cancer Incidence

$$\text{Incidence} = \underset{(\text{person-mg/kg/day})}{\text{Aggregate Dose}} \times \underset{(\text{mg/kg/day})^{-1}}{\text{Cancer Unit Risk Factor}}$$

$$\underset{(\text{person-mg/kg/day})}{\text{Aggregate Dose}} = \underset{(\text{person-}\mu\text{g/m}^3)}{\text{Aggregate Exposure}} \times \underset{\left(\frac{\text{mg-m}^3}{\mu\text{g-kg-day}}\right)}{\text{Inhalation Conversion Factor}}$$

$$\text{Inhalation Conversion Factor} = \frac{1\ \text{mg}}{1000\ \mu\text{g}} \times \frac{22.8\text{m}^3}{\text{person-day}} \times \frac{1\ \text{person}}{70\ \text{kg}} \times \frac{\%\ \text{Retention}}{100}$$

B. Calculation of Risk at Maximum Concentration Location

$$\text{Maximum Risk} = \underset{(\text{mg/kg/day})}{\text{Maximum Dose}} \times \underset{(\text{mg/kg/day})^{-1}}{\text{Cancer Unit Risk Factor}}$$

$$\underset{(\text{mg/kg/day})}{\text{Maximum Dose}} = \underset{(\mu\text{g/m}^3)}{\text{Maximum Concentration}} \times \underset{\left(\frac{\text{mg-m}^3}{\mu\text{g-kg-day}}\right)}{\text{Inhalation conversion factor}}$$

C. Calculation of Risk to Most Exposed Individual (MEI)

$$\text{MEI Risk} = \underset{(\text{mg/kg/day})}{\text{MEI Dose}} \times \underset{(\text{mg/kg/day})^{-1}}{\text{Cancer Unit Risk Factor}}$$

$$\text{MEI Dose} = \underset{(\text{mg/m}^3)}{\text{MEI Concentration}} \times \underset{\left(\frac{\text{mg-m}^3}{\mu\text{g/kg/day}}\right)}{\text{Inhalation Conversion Factor}}$$

Exhibit D-7

CARCINOGEN UNIT RISK FACTORS (1/(mg/kg/day))

EDC	6.90E-2
PCB	4.34E0
Chloroform	7.00E-2
2,3,7,8 - TCDD	1.56E+5
Arsenic	1.50E+1
Cadmium	7.80E0
Chromium	4.10E+1
Nickel	1.15E0

Source: *Health Assessment Document for Polychlorinated Dibenzo-p-Dioxins (Review Draft)*, EPA, May 1984.

Exhibit D-8

INHALATION HEALTH RISK ESTIMATES

	Texas		Arkansas	
	PCB	EDC	PCB	EDC
Cases				
Stack POHCs	3.32E-3	9.24E-3	5.50E-4	1.53E-3
Stack PICs	1.16E-1	1.68E-3	1.92E-2	2.77E-4
Metals	3.01E-1	3.68E-1	2.66E-1	3.25E-1
Fugitives	6.13E-3	1.38E-4	4.22E-3	9.64E-5
Total	4.27E-1	3.79E-1	2.90E-1	3.27E-1
Average Individual Risk				
Stack POHCs	1.31E-9	3.65E-9	6.06E-9	1.68E-8
Stack PICs	4.59E-8	6.60E-10	2.12E-7	3.05E-9
Metals	1.19E-7	1.46E-7	2.93E-6	3.58E-6
Fugitives	2.43E-9	5.53E-11	4.64E-8	1.06E-9
Total	1.69E-7	1.50E-7	3.20E-6	3.60E-6
Maximum Individual Risk				
Stack POHCs	6.10E-8	1.70E-7	4.16E-8	1.16E-7
Stack PICs	2.13E-6	3.07E-8	1.45E-6	2.10E-8
Metals	2.14E-5	2.61E-5	2.98E-5	3.63E-5
Fugitives	1.39E-7	2.96E-9	1.27E-6	3.09E-8
Total	2.37E-5	2.63E-5	3.25E-5	3.65E-5
Maximum Concentration Risk				
Stack POHCs	2.92E-7	8.10E-7	5.32E-8	1.47E-7
Stack PICs	1.02E-5	1.47E-7	1.86E-6	2.68E-8
Metals	2.16E-4	2.64E-4	3.76E-5	4.59E-5
Fugitives	7.31E-6	1.86E-7	1.55E-6	3.77E-8
Total	2.34E-4	2.65E-4	4.11E-5	4.61E-5

Source: IEc Analysis

Exhibit D-9

INHALATION RISK ESTIMATES FOR METALS

	Texas		Arkansas	
	PCB	EDC	PCB	EDC
Cases				
Arsenic	1.81E-1	2.21E-1	1.60E-1	1.95E-1
Cadmium	1.88E-2	2.30E-2	1.66E-2	2.03E-2
Chromium	9.89E-2	1.21E-1	8.74E-2	1.07E-1
Nickel	2.77E-3	3.39E-3	2.45E-3	2.99E-3
Total	3.01E-1	3.68E-1	2.66E-1	3.25E-1
Average Individual Risk				
Arsenic	7.15E-8	8.74E-8	1.76E-6	2.15E-6
Cadmium	7.44E-9	9.09E-9	1.83E-7	2.24E-7
Chromium	3.91E-8	4.78E-8	3.62E-7	1.18E-6
Nickel	1.10E-9	1.34E-9	2.70E-8	3.30E-8
Total	1.19E-7	1.46E-7	2.93E-6	3.58E-6
Maximum Individual Risk				
Arsenic	1.28E-5	1.57E-5	1.79E-5	2.18E-5
Cadmium	1.33E-6	1.63E-6	1.86E-6	2.27E-6
Chromium	7.01E-6	8.56E-6	9.76E-6	1.19E-5
Nickel	1.97E-7	2.40E-7	2.74E-7	3.34E-7
Total	2.14E-5	2.61E-5	2.98E-5	3.63E-5
Maximum Concentration Risk				
Arsenic	1.30E-4	1.58E-4	2.26E-5	2.76E-5
Cadmium	1.35E-5	1.65E-5	2.35E-6	2.87E-6
Chromium	7.08E-5	8.65E-5	1.23E-5	1.51E-5
Nickel	1.99E-6	2.43E-6	3.46E-7	4.23E-7
Total	2.16E-4	2.64E-4	3.76E-5	4.59E-5

Source: IEc Analysis

Exhibit D-10

RELATIVE DOSES FROM INHALATION AND INGESTION

Site	PCB Ingestion Dose (ug/yr)	PCB Inhalation Dose (ug/yr)	PCB Ingestion/ Inhalation Ratio	EDC Ingestion Dose (ug/yr)	EDC Inhalation Dose (mg/kg/d)	EDC Ingestion/ Inhalation Ratio
Northern Midwest				1.77E-2	3.34E-1	5.30E-2
Central Midwest	1.60E-3	2.30E-2	6.96E-2	1.18E-2	1.29E-1	9.15E-2
Southern California				4.25E-3	7.23E-1	1.91E-2
Average	1.60E-3	2.30E-2	6.96E-2	1.13E-2	2.29E-1	5.45E-2

Source: Multiple Pathways Screening Level Assessment of a Hazardous Waste Incineration Facility (Draft), G.A. Holton, C.C. Travis, E.L. Etnier, F.R. O'Donnel, D.M. Hetrick and E. Dixon, Oak Ridge National Laboratory, April 1984, and A Comparison of PCB Exposures to Human Populations from Oceanic and Terrestiral Incineration, G.A. Holton, C.C. Travis, E.L. Etnier, Life Systems, Inc., 1984.

Exhibit D-11

COMPARISON OF INHALATION AND INGESTION RISKS

	INCIDENCE				AVERAGE RISK			
	Texas		Arkansas		Texas		Arkansas	
	PCB	EDC	PCB	EDC	PCB	EDC	PCB	EDC
Ingestion	2.31E-4	5.03E-4	3.83E-5	8.33E-5	9.15E-11	1.99E-10	4.22E-10	9.17E-10
Inhalation	4.27E-1	3.79E-1	2.90E-1	3.27E-1	1.69E-7	1.50E-7	3.20E-6	3.60E-6
Total	4.27E-1	3.80E-1	2.90E-1	3.27E-1	1.69E-7	1.50E-7	3.20E-6	3.60E-6
Ingestion/Total (%)	.05	.13	.01	.03	.05	.13	.01	.03

Exhibit D-12

SUMMARY OF LIFETIME RISK ESTIMATES FOR HIGHEST RISK POLLUTANTS (Average Lifetime Risks)

	NESHAPS	Ambient Air Data	35 County Study
Formaldehyde	7.0E-7	5.8E-5	1.5E-5
Benzene	9.8E-6	7.6E-5	2.7E-5
Chromium	1.0E-4	7.4E-5	2.0E-5
Cadmium	4.9E-6	4.2E-6	1.4E-6
Nickel	2.5E-5	4.9E-6	1.4E-6
Arsenic	1.4E-6	1.8E-5	1.4E-6
Trichloroethylene	2.8E-6	7.7E-6	1.1E-5
Perchloroethylene	7.0E-7	7.7E-6	9.8E-6
Ethylene dichloride	1.3E-5	N/A	2.1E-6
Ethylene oxide	1.5E-5	N/A	N/A
Carbon tetrachloride	4.2E-6	2.6E-5	2.8E-7
Ethylene dibromide	8.4E-6	N/A	1.4E-6
Chloroform	<7.0E-7	3.2E-5	1.4E-7
Gasoline vapors	N/A	N/A	1.1E-5
All Other	7.0E-6	9.8E-6	2.4E-5
PIC	2.5E-4*	2.4E-4*	2.2E-4
Radionuclides	5.6E-6*	5.6E-6*	5.6E-6*
Asbestos	3.5E-5*	3.5E-5*	3.5E-5*
Gasoline Marketing	1.4E-5*	1.4E-5*	
Total	5.0E-4	6.1E-4	3.9E-4

* Risk estimates from other EPA efforts.

Source: Preliminary Draft EPA staff report titled, The Magnitude and Nature of the Air Toxics Problem in the United States, E. Haemisegger, A. Jones, B. Steigerwald, V. Thomson, 1984.

Exhibit D-13

INHALATION RISK COMPARISON

	Maximum Individual Risk	
	PCB Waste	EDC Waste
Texas	2.37E-5	2.63E-5
Arkansas	3.25E-5	3.65E-5
Air Toxics Report		
NESHAPS Study		4.1E-4
35 County Study		4.0E-4
Ambient Air Data		6.3E-4

Source: IEc Analysis

Appendix E

ESTIMATION OF PIC EMISSIONS FROM OCEAN- AND LAND-BASED INCINERATORS

This appendix describes the methodology and data used to estimate emission rates for products of incomplete combustion (PICs) from ocean- and land-based incinerators. In general, we define PICs to be compounds emitted in the incinerator stack gases which are not present in the waste itself. These may include actual products of incomplete combustion, fuel combustion products, or compounds volatilized from scrubber water. To compare risks from ocean- and land-based incineration, we estimated PIC emission rates for:

1. Land-based incineration of non-PCB organic wastes,
2. Land-based incineration of PCB wastes,
3. Ocean-based incineration of non-PCB organic wastes, and
4. Ocean-based incineration of PCB wastes.

Emissions estimates for the four cases listed above are based on PIC emission rates for these wastes as measured in trial burns at ocean- and land-based facilities. Emission rates for incineration of non-PCB organic compounds are calculated as the average over all trial burns (either on land or at sea) of the measured PIC emissions. These are expressed as a percent of the waste feed rate to make results from different test burns comparable.

For PCB wastes, the PIC emission rate used in our analysis is the average emission rate for tetrachlorodibenzo-p-dioxins (TCDDs), again expressed as a percent of the waste feed rate. We used this value instead of a broader measure of PICs (i.e. total organic PICs in the stack gases) primarily because data on other PICs were not available for the PCB trial burns. In the PCB trial burns at facilities where the data were most complete, the stack gases were sampled only for dioxin and furan PICs. In the trial burns aboard the Vulcanus, investigators sampled only for TCDD PICs. Analyses of stack gases were limited to these compounds because TCDDs are thought to be the most toxic PICs emitted during incineration of PCB wastes. To estimate the risks due to PIC emissions from PCB incineration we followed EPA's recommendations in using only TCDD emissions to evaluate the risks from emissions of dioxin and furan mixtures. (personal communication, Dr. Donald Barnes, EPA Office of Toxic Substances).

Overall, the PIC estimates presented in this appendix should be viewed as rough approximations of likely PIC emissions at the ocean- and land-based incinerators on which our comparative risk analysis is based. There are five major reasons for the imprecise nature of the PIC estimates, each of which is outlined below. First, the PIC emission rates found in the trial burns are based on differing definitions of what constitutes a PIC. In some trial burns PICs are defined as compounds not found in the waste itself while in other burns PICs are compounds not detected in the waste in concentration greater than 100 ppm. As a result, "PICs" are not always products of incomplete combustion.

A second factor complicating the estimation of PIC emissions is that no consistent set of compounds was considered in the trial burns. In the PCB trial burns, investigators generally looked only for dioxin and furan PICs. In the non-PCB burns, a broader group of PICs was sampled for, but the actual list of compounds for which the stack gases were analyzed was not the same in all the burns. Further, no consistent sampling methodology was used in the trial burns. Because the two major sampling trains used in the trial burns are sensitive to different compounds, the results may not be directly comparable between burns. Overall, given the limited number of compounds for which investigators analyzed the stack gases, it is likely that we have not accounted for all the PICs emitted from the trial burn incinerators.

A third factor reducing the accuracy of our PIC estimates is that the trial burn emission estimates are not based on a consistent set of wastes. For the land-based incinerator trial burns, a different waste stream was incinerated at each facility. For test burns aboard the Vulcanus, PIC data are available for three waste streams. None of the wastes on land or sea is identical to the wastes on which our analyses are based. Since PIC generation is relatively waste dependent, using emission rates based on other wastes streams may introduce additional error.

A fourth factor increasing the uncertainty associated with our estimates of PIC emission rates for the land-based units is that the emission rates are based on several types of incinerators each having somewhat different operating parameters. Since the generation of PICs is believed to be partially a function of the incinerator type, configuration, and operating conditions, emission rates based on facilities different from the ones assumed in our analysis may not be strictly correct.

Finally, a fifth source of uncertainty is introduced into our emission rate estimates as a result of disagreement about the validity of the results from the trial burns. For some of the burns aboard the Vulcanus, incorrect sampling protocols may have been followed, resulting in possible contamination of the samples with compounds which are neither PICs nor POHCs. This may result in an overstatement of the quantity of PICs.

With these five important caveats in mind, we summarize the results of our analysis of the trial burn data in Exhibit E-1. These data indicate that more PICs are produced when burning PCB wastes at land-based facilities than would result from incineration of similar wastes at sea. At the same time, however, the results suggest that PIC generation from burning non-PCB organic wastes at sea would be greater than would result from the incineration of similar wastes at land-based units. At present, we are not able to explain why ocean-based incineration generates relatively more PICs than land-based operations for some compounds and relatively fewer for others. Given this anomaly and the five sources of uncertainty outlined above, our estimates of PIC emission rates and the resulting estimates of risks from inhalation of PICs should be viewed only as rough approximations based on the best currently available data.

ESTIMATING PIC EMISSION RATES AT LAND-BASED INCINERATORS BURNING NON-PCB ORGANIC WASTES

We estimated PIC emission rates for land-based incinerators burning non-PCB organic waste streams using data developed from EPA sponsored trial burns at eight land-based incinerators (see Performance Evaluation of Full-Scale Hazardous Waste Incinerators Midwest Research Institute, Kansas City, Missouri, 1984) Although different wastes were burned in each of the trial burns, all eight waste streams were comprised principally of organic compounds other than PCBs.

In performing the trial burns at the land-based incinerators, MRI used the VOST procedure to sample the stack gases for volatile organic PICs. (For a more detailed description of sampling techniques, see "Sampling and Analysis Methods for Hazardous Waste Combustion, A.D. Little, Inc., February, 1983.) PICs were defined as any compound emitted in the stack gases which was not present in the waste in concentrations greater than 100 ppm. As a result of the 100 ppm cutoff value, some of the PICs found in the stack gas samples are likely to be compounds which were actually present in the waste ifself and not products of incomplete combustion.

To identify PICs, MRI analyzed the gas samples for RCRA Appendix VIII pollutants using gas chromatography/mass spectroscopy (GC/MS) techniques. Because the stack gas samples were analyzed only for these pollutants, basing our estimates of PIC emission rates on these analyses may tend to understate the total quantity of PICs. Estimates based on the MRI data will also tend to understate the quantity of PICs because only about half of the Appendix VIII pollutants can be identified using the GC/MS techniques employed by MRI (personal communication, A. Trenholm, MRI, October 1984).

Although the total quantity of PICs is not known with certainty for these trial burns, MRI investigators have estimated that, on average, the measured PICs account for approximately one percent of the total unburned hydrocarbon emissions (personal communication, A. Trenholm, MRI, October 1984). Some of the unburned hydrocarbons would be POHC emissions, but some are likely to be PICs for which the stack gas samples were not specifically analyzed. Although we have not done so explicitly in our analyses, an upper bound on the total PIC emissions could be established by assuming that all the unburned hydrocarbons are PICs.

Emission rates for PICs are reported in Exhibit E-2 for each of the eight trial burns. The MRI report provides actual emission rates accompanied by blank correction factors for each site and compound detected. These correction factors are the mean and standard deviation of the compound levels for traps (known as "blanks") which were exposed to the same conditions as the sampling traps, but were not used in the actual sampling process. The purpose of using blanks is to detect and correct for contamination of the sampling traps. Blank correction of the data was performed by IEc using the following procedure:1/

1. For each sample/compound combination, the sample value and the average blank value we compared to determine if they are significantly different. A sample value is "different" from the average blank value if,

   Sample Value (ng) - Average Blank Value (ng) $\geq$

   3 x Blank Standard Deviation (ng)

2. If the sample value is determined to be "different" than the average blank value, the emission value is calculated in the following manner,

   Emission Value (ng) =

   Sample Value (ng) - Average Blank Value (ng)

3. If the sample value is not determined to be "different" from the average blank value, the sample value is treated as an upper bound on the emission value, i.e.,

   Emission Value $\leq$ Sample Value.

A total of thirty-five PICs were detected in the stack gases. While the PIC emissions are sensitive to the constituents of the waste stream, some chemicals appear more consistently as PICs than others. Benzene and chloroform are detected in five of the eight burns.

1/The discussion of blank correction is drawn from "Performance Evaluation of Full-Scale Hazardous Waste Incinerators", Volume II, pp. 116-118.

At the bottom of Exhibit E-2, the measured PIC emission rate is calculated for each of the trial burns by summing the emissions for each of the individual compounds detected in the stack gases. These emission rates, however, are not directly comparable between trial burns since each burn is based on a different waste feed rate. Waste feed rates are reported at the bottom of Exhibit E-2. To make the emission rates directly comparable between burns , we re-calculated PIC emissions as a percent of the original waste feed rate by dividing the emission values in Exhibit E-2 by the waste feed rate for each of the trial burns. These values are presented in Exhibit E-3.

Measured PIC emissions as a percent of the waste feed rate are calculated for each of the trial burns by summing the emissions for each of the compounds detected. These values range from 1.01E-5 to 5.26E-3 percent of the waste feed rate. The mean of the measured PIC emission rates is calculated by averaging the totals for the eight trial burns, and it is this average rate which is used for all calculations of risks from PIC emissions at land-based incinerators burning non-PCB organic wastes. This average (8.94E-4 percent) suggests that on a total mass basis, PICs are emitted at a rate which is less than 1/10th of the allowable rate for POHCs. Had we assumed that the total unburned hydrocarbon emissions are a better estimate of PIC emission rates, our estimates would increase by approximately two orders of magnitude, resulting in PIC emissions approximately 9 times higher than POHC emissions.

In general, this averaging procedure produces only rough estimates of the average PIC emission rates from these incinerators. The trial burns took place at eight different facilities with different types of incinerators and operating characteristics. PIC formation is believed to be a function of the incinerator type and operating parameters. Averaging across a range of facility types and operating conditions will not necessarily produce the appropriate estimate of PIC emissions for the incinerators on which our comparative risk assessment is based. However, because emission rates for the eight burns differ only by one to two orders of magnitude, we have more confidence that the resulting PIC emission rates are approximately correct.

ESTIMATING PIC EMISSION RATES AT LAND-BASED INCINERATORS BURNING PCB WASTES

To estimate PIC emission rates from the burning of PCB wastes at land-based incinerators, we analyzed data on PICs from seven PCB trial burns. Five of the burns were conducted at the SCA Chemical Services, Inc. rotary kiln incinerator in Chicago, and single trial burns were performed at the Energy Systems Company (Ensco) facility in El Dorado, Arkansas, and the Rollins Environmental Services multiple chamber incinerator in Deer Park, Texas. (For more detailed reports of results from these trial burns see _An Air Resources Board Policy Regarding Incineration as an Acceptable Technology for PCB Disposal_, State of California Air Resources Board, December 1981, and _Environmental Risk Assessment for the Incineration of Polychlorinated Biphenyls_, Charles T. Main Inc., March 1983). All three of these units are currently permitted to burn PCB wastes.

At each of the three facilities, EPA's Modified Method 5 sampling procedure was employed. In the SCA trial burns the stack gases were analyzed for polychlorinated dibenzodioxin (PCDD) and polychlorinated dibenzofuran (PCDF) PICs. For the Ensco and Rollins trial burns, stack gases were analyzed only for TCDD and TCDF PICs, a subset of the total PCDDs and PCDFs. Emission rates for other organic PICs could not be determined since the samples were not analyzed for these compounds at any of the three facilities. Thus, the estimated quantities of dioxins and furans will understate the total quantities of PICs emitted in the stack gases. Since data on total unburned hydrocarbon emissions were not available for these trial burns, it was not possible to perform rough calculations to determine a worst case estimate of the percentage of the total PIC emissions represented by the measured PICs (dioxins and furans).

PIC emission rates by compound are summarized in Exhibit E-4 for each of the seven trial burns. As noted above, stack gases were sampled only for dioxins or furans. The emission rates in Exhibit E-4 are not directly comparable between test burns since a different waste feed rate was used in each of the burns. To put them on a comparable basis, we calculated the PIC emissions as a percentage of the waste feed rate. These values are presented in Exhibit E-5.

Exhibit E-5 indicates that total PIC emissions summed over all the detected dioxins and furans ranged from 3.78E-9 to 7.77E-7 percent of the waste feed rate. However, since EPA recommends that analysis of risks from mixtures of the various types of dioxins be based only on the TCDD emissions, we used the average TCDD emission rate (3.40E-8) as the percentage of PICs emitted during incineration of the PCB waste streams (personal communication, Dr. Donald Barnes, EPA Office of Toxic Substances, September 1984).

ESTIMATING PIC EMISSION RATES FROM OCEAN-BASED INCINERATORS BURNING NON-PCB ORGANIC WASTES

PIC emission rates for ocean-based incineration of non-PCB organic wastes were estimated based on data from trial burns of non-PCB containing organic wastes aboard the Vulcanus (reported in History of Environmental Testing of the Chemical Waste Incinerator Ships M/T Vulcanus and I/V Vulcanus II, D.G. Ackerman, J.F. Metzger, and L.L. Scinto, TRW Inc., Redondo Beach, CA, June 1983). Data to analyze PICs were available only from the Shell II and Herbicide Orange burns aboard the Vulcanus. For each of the wastes three trail burns were performed.

Two similar sampling methods were selected for the Herbicide Orange and Shell II trial burns. In the Herbicide Orange case, EPA's Modified Method 5 sampling train was employed, while in the Shell II burns, the SASS sampling train was used. In the SASS method, a larger volume of gas is sampled, resulting in a shorter sampling period. Otherwise, the sampling procedures are identical.

For the Herbicide Orange trial burns, GC/MS techniques were used to analyze the stack gases for volatile organic compounds having between seven and sixteen carbon atoms. Any PICs with fewer than six carbon atoms would not have been detected. Individual compounds were not identified, and as a result it was only possible to estimate the total quantity of C7 through C16 PICs. In addition, the SASS sampling methodology may not have captured the most volatile organic compounds (compounds for which the VOST sampling technique used in the MRI trial burns is more effective). As a result, the measured quantity of PICs may also understate the total quantity emitted from the stack of the Vulcanus. Relative to the total unburned hydrocarbons emitted in the stack gases, measured PICs account for less than one percent of the total hydrocarbon emissions. Again, the reader should note that an absolute worst case estimate of PIC emissions could be made by assuming that all the unburned hydrocarbons are PICs. In this case, such an assumption would imply PIC emission rates approximately two orders of magnitude greater than the ones used in our analyses.

For the Shell II burns, the stack gases were analyzed for a range of volatile species using GC/MS techniques. As was the case in the Herbicide Orange tests, highly volatile organic compounds might not have been captured by the SASS sampling train and thus estimates of PIC emissions may be understated. A comparison of the measured PIC emissions with the total unburned hydrocarbons emitted from the stack indicates that measured PICs account for less than 0.1 percent of the unburned hydrocarbons. Thus, a worst case estimate of PIC emissions might be as much as three orders greater than the number estimated in our analysis.

The methodology used to estimate PIC emissions from the ocean-based incinerator is identical to the method described for the land-based units. Exhibit E-6 shows PIC emissions for the six test burns. Because data for the Herbicide Orange trial burns were not analyzed for specific constituents, only total measured PIC emissions are reported for these burns. As before, PIC emissions are standardized to reflect emissions as a percent of the initial waste feed rate; these values are presented in Exhibit E-7. The average PIC emission rate (3.01E-2) is the mean of the total emission rates for the six trial burns. This average emission rate is approximately 34 times higher than the PIC emission rate (8.94E-4) for land-based incinerators burning non-PCB organic wastes. The reason for the higher emission rate aboard the Vulcanus is unknown. Some of the difference may be attributable to improper testing protocols used in the Shell and Herbicide Orange trial burns. In these tests, the resins used to trap the organic compounds may have overheated causing resin breakdown products to form. Although some of the PICS identified may be resin breakdown products, they are unlikely to account for a large part of the difference between the ocean and land emission rates (personal communication, Dr. Judith Harris, A.D. Little, Inc.) Other factors which might contribute to the higher PIC rate for the Vulcanus include possible differences in the wastes burned in the land and ocean trial burns, differences in the compounds for which the stack gases were analyzed, and perhaps differences in the residence time and temperature at which the land and ocean-based units operate.

ESTIMATION OF PIC EMISSION RATES FOR OCEAN-BASED INCINERATORS BURNING PCB WASTES

The estimation of PIC emission rates for ocean incineration of PCB wastes is based on data published in At-Sea Incineration of PCB-Containing Wastes Onboard the M/T Vulcanus (EPA Office of Water, April 1983). EPA researchers analyzed the stack gases from ten trial burns for TCDDs and TCDFs. Stack gases where sampled using the Modified Method 5 sampling train. Samples were analyzed for TCDDs and TCDFs using GC/MS techniques. As was true for the PCB trial burns at the land-based incinerators, stack gases were not sampled for a broad range of organic compounds. As a result, the estimate of emissions used in our analyses will not reflect the total quantity of PICs emitted from an incinerator ship. This may result in a significant understatement of the total PIC emissions from the incinerator. Because total unburned hydrocarbon emissions were not monitored during these trial burns, it is not possible to estimate measured PICs as a proportion of the total unburned organic species emitted in the stack gases.

The results of our analysis of PIC emissions from the PCB waste trial burns aboard the Vulcanus are shown in Exhibits E-8 and E-9. No TCDDs were detected, and while some TCDFs were found, we have assumed that these were POHCs since they were also found in the waste itself.

Although no TCDDs were detected in the analysis of the incinerator emissions, this does not necessarily mean that no TCDDs were present. The testing procedures indicated that TCDDs should have been detected in a sample only if they were present in quantities greater than a minimum detection limit. The detection limits ranged from 0.04 to 0.53 nanograms for the various stack gas samples. Based on these detection limits, we calculated the maximum amount of TCDD which could exist in the waste, but which would not be detected in the analyses of the emissions. We performed this calculation for each trial burn by determining the sum of the detection limits for the Floracil and probe and impinger tests and dividing this by the amount of waste burned during the sampling procedure. For the test burn with the highest detection limits (i.e., the test where the greatest amount of TCDD could have been present and not detected), TCDDs could have been present in the stack emissions in quantities less than or equal to 1.1E-11 percent of the waste feed and not have been detected. Since it is impossible to say that no TCDD PICs were present in the stack gases and because these compounds are highly carcinogenic, we made the conservative assumption that TCDDs were present in the stack gases at the minimum detection level.

As noted earlier, even though no TCDDs were detected in the stack gas analyses for the trial burns, this result is probably not an accurate predictor of the total PIC emissions from the PCB burns since only TCDDs and TCDFs were sampled for. Other PICs could have been present in the stack gases. The results from the PCB burns at the land-based incinerators indicate that various other dioxins and furans can be present in the PCB waste combustion gases. In addition, numerous other hazardous and non-hazardous organic compounds are likely to be present if the more comprehensive analyses performed for trial burns of non-PCB wastes are indicative of the types of emissions from burns of PCB wastes. Therefore, although we have accounted for the most toxic PICs emitted during incineration of PCB wastes, we have not accounted for what is probably a large volume of less toxic substances.

ASSESSING THE RELIABILITY OF THE PIC EMISSION RATE ESTIMATES

Although estimated using the best available data, the PIC emission rates described in this appendix are subject to considerable uncertainty. As described previously, five major factors contribute to this uncertainty. First, some of the trial burns use differing definitions of PICs. In the MRI trial burns of non-PCB organic wastes at land-based incinerators, PICs are defined to be any compound present in the stack gases which was not in the waste at concentrations greater than 100 ppm. In the trial burns of PCBs on land and in burns of both wastes aboard the Vulcanus, PICs were defined to be compounds not present in the waste above the detection limits for the tests employed. Since these limits vary from one set of trial burns to another, the definition of PICs is not consistent across burns. This reduces the reliability of estimates derived by averaging PIC emissions across burns with different operative definitions of PICs. This problem arises in the estimates of emission rate estimates for PCB wastes at land-based facilities and non-PCB wastes burned at sea. Average emission rates for these wastes are based on burns conducted using different sampling procedures and detection limits. For the other two emission rate estimates (non-PCB wastes on land and PCB wastes at sea) the definition of a PIC is internally consistent.

The second factor contributing to the uncertainty of the PIC emission rates developed from the trial burn data is that no consistent list of compounds or sampling procedures was used to analyze stack emissions of PICs. In the MRI trial burns, investigators looked for RCRA Appendix VIII pollutants. In the trial burns of non-PCB organic wastes aboard the Vulcanus, stack gases were analyzed for a range of organic compounds. Because these samples were not analyzed in all cases for specific compounds, it is not possible to determine the extent of their overlap with the Appendix VIII compounds sampled for at the land-based trials.

For the PCB test burns at both ocean- and land-based facilities, investigators attempted to quantify only dioxin and furan PICs. In the burns at the SCA incinerator in Chicago, they analyzed the stack gases for PCDDs and PCDFs. At the Ensco and Rollins units and aboard the Vulcanus, the stack gases were analyzed only for TCDDs and TCDFs, a limited subsets of the total PCDD and PCDF emissions. Thus, these results are comparable only on the basis of the TCDD and TCDF emissions.

Further complicating the analysis of average PIC emission rates is the fact that sampling procedures between trial burns were not always consistent. In general, two sampling trains were employed, the VOST and the Modified Method 5. The VOST procedure was used for the MRI trial burns, and is an effective technique for sampling highly volatile organic compounds. The Modified Method 5, used in all other tests except the Shell II burns, is an efficient technique for capturing all but the most volatile compounds. For the highly volatile species, the VOST will provide more efficient collection. Although the choice of sampling train should not affect the results from the PCB trial burns where only the dioxins and furan were considered, it probably makes a difference in the non-PCB burns. Because the VOST method was used in the analyses of the trial burns at land-based facilities, emissions rates estimated for these facilities may be more accurate than those from the Herbicide Orange and Shell II burns aboard the Vulcanus.

In addition to inconsistencies in sampling methodologies and selection of compounds, PIC estimates based on the trial burn data in general are likely to underestimate the total quantities of PIC emissions. In none of the trial burns were all the organic PICs in the stack gases analyzed. In cases for which data were available, measured PICs accounted for less than one percent of total unburned hydrocarbon emissions which are composed of both PICs and POHCs. As a result, we believe the estimates of PIC emissions developed in this analysis probably understate the total PIC emissions. This is especially likely to be the case for the PCB wastes where only TCDD PICs are accounted for.

The third factor reducing the reliability of our estimates of PIC emission rates is that no consistent set of wastes was burned in all the trial burns. As a result, estimates of individual PIC emission rates are not based on the same waste stream. This was true for the non-PCB organics waste trial burns on land and at sea and for the PCB burns at land-based units. Since PIC generation is believed to be at least partially waste dependent, single estimates based on a variety of waste streams may be only rough approximations of PIC emission rates. However, since the PIC emission rates for individual trial burns tend to differ by only one of two orders of magnitude, we believe basing our PIC emission rate on multiple waste streams provides a reasonable estimate.

The fourth factor contributing to the uncertainty of the PIC estimates results from basing the PIC emission rates for land-based facilities on trial burns at different types of incinerators. The MRI trial burns were performed at both liquid injection and rotary kiln incinerators. Since PIC formation is partially dependent on the incinerator type and its operating conditions, basing emission rate estimates on a variety of facilities will not necessarily produce an appropriate estimate for an individual facility. However, because our comparative risk analysis is itself not based on any particular facility type or configuration, average PIC emission rates based on different facilities should provide a reasonable estimate of PIC emissions.

The final factor introducing uncertainty into our estimates of PIC emission rates is the possibility that sampling procedures used in the Shell II and Herbicide Orange trial burns abroad the Vulcanus may not result in accurate prediction of PIC generation rates. The resins on which the volatile organics compounds were collected during these burns may have been allowed to overheat, resulting in the formation of resin breakdown products which were erroneously identified as PICs. However, based on discussions with Dr. Judith Harris of Arthur D. Little, we believe resin breakdown products do not account for a significant portion of the PICs detected in these burns.

Overall, the estimate of PICs emissions rate developed from the trial burn data are rough approximations of the quantity of PICs emitted from incinerators. Although they do not account for all organic PICs, they probably are representative of the most hazardous compounds emitted. In the absence of better data, we believe the estimates described in this appendix provide the best available estimate of PIC emissions.

Exhibit E-1

AVERAGE PIC EMISSION RATES
(percent of waste feed)

	PCB Wastes	Non-PCB Organic Wastes
Land-Based	3.40E-8	8.94E-4
Ocean-Based	1.13E-11	3.01E-2

Source: IEc Analysis

Exhibit E-2

PIC EMISSION RATES
LAND BASED INCINERATORS
NON-PCB ORGANIC WASTES

PRODUCTS OF INCOMPLETE COMBUSTION	MRI STUDY BURNS (ug/min)							
	1	2	3	4	5	6	7	8
1,1,1-Trichloroethane				1.61E+01	2.53E+01	8.20E+00		
1,2,4-Trichlorobenzene		1.50E+03						
2,4,6-Trichlorophenol			5.07E+03					
2,4-Dimethyl phenol						6.30E+02		
2,6-Toluene diisocyanate			2.20E+02					
Benzene	9.00E+03		3.60E+03	5.64E+02	4.95E+02		2.47E+05	
Bromoform	8.15E+02						7.61E+01	3.46E+03
Carbon disulfide	2.67E+04							
Carbon tetrachloride					6.80E+01			
Chlorobenzene					7.90E+01	8.04E+01	1.92E+03	
Chloroform	7.29E+03		8.29E+04	5.95E+01	3.24E+02	1.00E+02		
Dibromochloromethane	4.83E+03		1.13E+03				3.64E+02	4.30E+02
Dichlorobromomethane	4.51E+03		2.20E+03				1.14E+03	5.77E+02
Diethyl phthalate			3.80E+02					
Dimethyl phthalate		1.70E+02						
Fluoranthene	1.07E+03							
Hexachlorobenzene		1.40E+03	2.27E+01					
Methyl bromide	1.23E+03							
Methyl chloride	4.05E+03							
Methyl ethyl ketone			1.72E+02					
Methylene bromide	1.54E+04							
Methylene bromochloride	1.60E+04							7.43E+02
Methylene chloride			8.01E+02			1.50E+02		
Naphthalene			2.64E+02		6.00E+03			1.93E+03
Pentachlorophenol			3.40E+02					
Phenol			2.27E+02				8.13E+03	
Pyrene	8.33E+02							
Tetrachloroethylene			1.53E+02	1.60E+01	1.22E+01			
Toluene			3.83E+01		2.36E+02			
Trichloroethylene					4.30E+02			
o-dichlorobenzene		8.43E+02						
o-Chlorophenol			4.75E+02					
o-Nitrophenol			1.34E+03		2.50E+03			
m-dichlorobenzene		1.07E+03						
p-dichlorobenzene		6.03E+02						
TOTAL	9.63E+04	5.67E+03	1.03E+05	6.56E+02	1.11E+04	9.69E+02	2.99E+05	7.14E+03
AVERAGE WASTE FEED RATE (kg/min)	98.767	41.140	1.963	0.743	2.522	9.635	22.573	7.263

SOURCE: GC analysis of trial burn data
Midwest Research Institute, "Performance Evaluation of Hazardous Waste Incinerators", 1984.

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Exhibit B-1

PIC EMISSION RATES (% of Waste Feed)
LAND BASED INCINERATORS
NON-PCB ORGANIC WASTES

PRODUCTS OF INCOMPLETE COMBUSTION	AVERAGE EMISSION RATE (% of Waste Feed)	MRI STUDY BURNS 1	2	3	4	5	6	7	8
1,1,1-Trichloroethane	4.07E-07				2.17E-06	1.00E-06	8.51E-08		
1,2,4-Trichlorobenzene	4.56E-07		3.65E-06						
2,4,6-Trichlorophenol	3.74E-05			2.99E-04					
2,4-Dimethyl phenol	8.17E-07						6.54E-06		
2,6-Toluene diisocyanate	1.27E-06			1.0?E-05					
Benzene	1.73E-04	9.??E-06		1.93E-04	7.40E-05	1.98E-05		1.09E-03	
Bromoform	7.03E-06	8.25E-06						3.37E-07	4.76E-05
Carbon disulfide	3.38E-06	2.70E-05							
Carbon tetrachloride	3.37E-07					2.70E-06			
Chlorobenzene	1.56E-06					3.11E-06	8.34E-07	8.51E-06	
Chloroform	5.31E-04	7.30E-06		4.22E-03	7.97E-06	1.20E-05	1.04E-06		
Dibromochloromethane	8.56E-06	4.00E-06		5.70E-05				1.61E-06	5.92E-06
Dichlorobromomethane	1.67E-05	4.57E-06		1.16E-04				5.05E-06	7.94E-06
Diethyl phthalate	2.42E-06			1.94E-05					
Dimethyl phthalate	5.17E-08		4.13E-07						
Fluoranthene	1.35E-07	1.08E-06							
Hexachlorobenzene	5.94E-07		3.64E-06	1.10E-06					
Methyl bromide	1.56E-07	1.25E-06							
Methyl chloride	5.13E-07	4.10E-06							
Methyl ethyl ketone	1.10E-06			8.78E-06					
Methylene bromide	1.95E-06	1.56E-05							
Methylene bromochloride	3.30E-06	1.62E-05							1.02E-05
Methylene chloride	5.29E-06			4.08E-05			1.56E-06		
Naphthalene	3.91E-05			1.35E-05		2.73E-04			2.66E-05
Pentachlorophenol	2.16E-06			1.73E-05					
Phenol	5.75E-06			1.16E-05				3.44E-05	
Pyrene	1.05E-07	8.43E-07							
Tetrachloroethylene	1.32E-06			7.71E-06	2.25E-06	6.0?E-07			
Toluene	2.09E-05			1.54E-04		9.41E-06			
Trichloroethylene	2.15E-06					1.72E-05			
m-dichlorobenzene	2.56E-07		2.05E-06						
o-Chlorophenol	3.05E-06			2.44E-05					
o-Nitrophenol	2.10E-05			6.83E-05		9.97E-05			
o-dichlorobenzene	3.25E-07		2.60E-06						
p-dichlorobenzene	1.83E-07		1.47E-06						
TOTAL	8.94E-04	3.95E-05	1.30E-05	5.25E-03	9.84E-05	4.37E-04	1.01E-05	1.15E-03	9.83E-05

SOURCE: ICF analysis of trial burn data
Midwest Research Institute, "Performance Evaluation of Hazardous Waste Incinerators", 1984.

Exhibit E-4

PIC EMISSION RATES
LAND BASED INCINERATORS
PCB WASTES

PRODUCTS OF INCOMPLETE COMBUSTION	SCA TRIAL BURNS					ROLLINS	ENSCO
	10 (ug/sec)	11 (ug/sec)	12 (ug/sec)	13 (ug/sec)	14 (ug/sec)	8/80 (gm/sec)	8/80 (gm/sec)
Octa CDD	4.00E-01	0.00E+00	1.00E-01	5.29E-01	2.70E-01		
Di CDF	2.67E-01	3.17E-01	0.00E+00	3.17E-01	1.05E+00		
Tri CDF	2.13E-01	0.00E+00	4.07E-02	3.50E-02	1.05E+00		
Octa CDF	0.00E+00	2.50E-02	0.00E+00	1.00E+00	1.04E-01		
TCDD	3.43E-02	2.13E-01	4.07E-02	1.63E-01	1.57E-01	1.03E-08	1.69E-09
Mono CDF	0.00E+00	3.65E-01	0.00E+00	0.00E+00	2.50E-01		
Pent CDF	2.67E-02	1.63E-01	3.13E-02	1.63E-01	4.67E-01		
TCDF	1.37E-02	2.50E-02	0.00E+00	2.07E-02	1.37E-01	1.60E-07	2.12E-08
Hexa CDF	0.00E+00	0.00E+00	0.00E+00	0.00E+00	1.00E+00		
Hepta CDF.	0.00E+00	0.00E+00	0.00E+00	5.29E-03	5.29E-02		
TOTAL	9.73E-01	1.05E+00	2.13E-01	2.10E+00	4.36E+00	1.70E-07	2.29E-08
	(g/min)	(g/min)	(g/min)	(g/min)	(g/min)	(lb/hr)	(lb/hr)
AVERAGE WASTE FEED RATE	7.53E+03	1.10E+04	4.00E+03	2.83E+04	4.90E+04	5.32E+03	4.00E+03

SOURCE: IEc analysis of trial burn data
SCA data from: "Environmental Risk Assessment for the Incineration of Polychlorinated Biphenyls" by Chas. T. Main, Inc, 3/8/83.
Rollins and Ensco Data from: "An Air Resources Board Policy Regarding Incineration as an Acceptable Technology for PCB Disposal"
State of California Air Resources Board: Stationary Source Control Division,
December 1981, pp. C-8, C-27.

Exhibit E-5

PIC EMISSIONS (% of Waste Feed)
LAND BASED INCINERATORS
PCB WASTES

PRODUCTS OF INCOMPLETE COMBUSTION	AVERAGE EMISSION RATE (% of Waste Feed)	SCA Trial Burns					ROLLINS	ENSCO
		10	11	12	13	14	8/80	8/80
Di CDF	1.19E-07	2.13E-07	1.62E-07	0.00E+00	9.37E-08	1.29E-07		
Hepta CDF	1.28E-09	0.00E+00	0.00E+00	0.00E+00	1.54E-11	6.37E-09		
Hexa CDF	2.43E-08	0.00E+00	0.00E+00	0.00E+00	0.00E+00	1.22E-07		
Mono CDF	4.36E-08	0.00E+00	1.86E-07	0.00E+00	0.00E+00	3.16E-08		
Octa CDD	1.26E-07	3.19E-07	0.00E+00	1.23E-07	1.54E-07	2.31E-08		
Octa CDF	6.41E-08	0.00E+00	1.28E-08	0.00E+00	2.96E-07	1.22E-08		
Pent CDF	4.00E-08	2.13E-08	5.26E-08	3.05E-08	3.05E-08	5.72E-08		
TCDD	3.40E-08	2.73E-08	1.09E-07	5.00E-08	3.05E-08	1.92E-08	1.54E-09	2.79E-10
TCDF	1.10E-08	1.11E-08	1.32E-08	0.00E+00	6.12E-09	1.92E-08	2.38E-08	3.58E-09
Tri CDF	7.50E-08	1.86E-07	0.00E+00	5.00E-08	1.06E-08	1.29E-07		
TOTAL (% of Waste Feed)	4.85E-07	7.77E-07	5.33E-07	2.61E-07	6.21E-07	5.59E-07	2.53E-08	3.78E-09

SOURCE: IEc analysis of trial burn data
SCA data from "Environmental Risk Assessment for the Incineration of Polychlorinated Biphenyls", Charles T. Main, Inc, 3/8/83.
Rollins and Ensco Data from: "An Air Resources Board Policy Regarding Incineration as an Acceptable Technology for PCB Disposal", State of California Air Resources Board: Stationary Source Control Division, December 1981, pp C-8, C-27.

Exhibit E-7

PIC EMISSIONS (% of Waste Feed)
OCEAN-BASED INCINERATORS
NON-PCB ORGANIC WASTE

		Shell Waste Burns			Herbicide Orange Burns		
PRODUCTS OF INCOMPLETE COMBUSTION	AVERAGE EMISSION RATE (% of Waste Feed)	IV	V	VI	1-ST-716-H 1-LR-716-H 1-PR-717-H	2-ST-813-H 2-LR-813-H 2-PR-813-H	3-ST-828-H 3-LR-828-H 3-PR-828-H
Butyl cresol	7.00E-05	2.10E-04	2.10E-08	2.10E-08			
Dichloro, c4, substituted benzene	2.80E-04	8.40E-04	9.10E-08	6.30E-08			
Dichloro, c5, substituted benzene	2.33E-05	7.00E-05	3.50E-08	1.40E-08			
Diethylbenzene	1.40E-04	4.20E-04	4.20E-08	4.20E-08			
Tetrachloro, c2, substituted benzene	1.40E-04	1.40E-04	1.40E-04	1.40E-04			
Tetrachlorobutene	1.63E-04	7.00E-05	3.50E-04	7.00E-05			
Tetrachloroethene	5.60E-05	6.30E-05	6.30E-05	4.20E-05			
Tetrahydronaphthalene	3.03E-05	3.50E-05	2.80E-05	2.80E-05			
Trichloro, alkyl substituted benzene	2.31E-04	2.10E-04	2.38E-04	2.45E-04			
Triethylbenzene	7.00E-05	7.00E-05	7.00E-05	7.00E-05			
TOTAL	3.01E-02	2.13E-03	8.09E-04	5.95E-04	4.25E-02	9.59E-02	3.08E-02

SOURCE: IEc analysis of trial burn data
Shell Burns: Addendum 2, Table 17 of "History of Environmental Testing of the Chemical Waste Incinerator Ships M/T Vulcanus and I/V Vulcanus II", TRW, Inc., June 1983.
Herbicide Orange Burns: Addendum 2, Table 13 of "History of Environmental Testing ...", June 1983.

Exhibit E-8

PIC EMISSION RATES
OCEAN BASED INCINERATORS
PCB WASTES

PRODUCTS OF INCOMPLETE COMBUSTION	AUGUST '82 BURNS 3 (ug/hr)	 6 (ug/hr)	 7 (ug/hr)	 8 (ug/hr)	 9 (ug/hr)	 10 (ug/hr)
TCDD	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00
TCDF *	(1.52E+02	(1.30E+01	(7.23E+01	(4.43E+01	6.01E+00	(2.03E+01
TOTAL	(1.52E+02	(1.30E+01	(7.23E+01	(4.43E+01	6.01E+00	(2.03E+01
	(mt/hr)	(mt/hr)	(mt/hr)	(mt/hr)	(mt/hr)	(mt/hr)
AVERAGE WASTE FEED RATE	6.04E+00	5.74E+00	4.98E+00	5.42E+00	5.38E+00	5.28E+00

* A "(" character means that TCDFs were not detected. The detection limit is used as a conservative maximum value possible. In addition, since TCDFs appear in the waste feed, they may not be PICS.

SOURCE: IEc analysis of trial burn data
Data from: "At-Sea Incineration of PCB-Containing Wastes Onboard the M/T Vulcanus"
by Industrial Environmental Research Laboratory, April 1983.

Exhibit E-9

PIC EMISSION RATES (% of Waste Feed)
OCEAN BASED INCINERATORS
PCB WASTES

PRODUCTS OF INCOMPLETE COMBUSTION	AVERAGE EMISSION RATE (% of Waste Feed)	AUGUST '82 BURNS 3	6	7	8	9	10
TCDD	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00
TCDF *	(3.14E-12	(2.52E-12	(2.25E-13	(1.40E-11	(6.17E-13	1.28E-13	(3.04E-13
TOTAL	(3.14E-12	(2.52E-12	(2.25E-13	(1.40E-11	(6.17E-13	1.28E-13	(3.04E-13

* A "(" character means that TCDFs were not detected. The detection limit is used as a conservative maximum value possible. In addition, since TCDFs appear in the waste feed, they may not be PICs.

SOURCE: ICF analysis of trial burn data
Data from: "At-Sea Incineration of PCB-Containing Wastes Onboard the M/T Vulcanus"
by Industrial Environmental Research Laboratory, April 1983.

Appendix F

ANALYSIS OF SCRUBBER EFFLUENT FROM LAND-BASED INCINERATORS

INTRODUCTION

This Appendix provides information on the generation and disposal of scrubber effluent from land-based incinerators.

PROCESS DESCRIPTION

The incineration process includes the entire chain of steps beginning with the introduction of wastes into the incinerator and ending with the disposition of scrubber effluents. The process is depicted in Figure F-1. The incinerator is assumed to be a liquid-burning type of equivalent performance to the ship-borne units. The principal existing commercial land-based facilities incorporate a rotary-kiln which, in turn, exhausts into an afterburner to which liquid wastes can be fed (Point 1). Liquids can also be introduced directly into the kiln portion.

The incinerator is assumed to destroy and remove 99.9999 percent of PCB containing wastes and 99.99 percent of other wastes.

The exhaust gases are quenched by direct contact with a water spray. The cooled gases then are scrubbed to remove particulates and acid gases such as hydrogen chloride. A wide variety and combination of equipment types can be used. For a purely liquid-burning facility, particulate loading in the exhaust gases may be quite low compared to a rotary kiln burning solids. In these cases, requirements for the particulate removal device will be moderate. In rotary kilns and in liquid burning units fed with high solids-containing feeds, high efficiency devices will have to be used to meet the stringent emission standards generally applied to hazardous waste incinerators. The standard is typically around 0.08 grains per dry standard cubic foot.

The gaseous contacting unit must meet federal and state requirements for chlorine removal if halogenated hydrocarbons are fed in quantities releasing four pounds per hour of chlorine or more. The minimum required is 99 percent removal, which figure is assumed in this analysis.

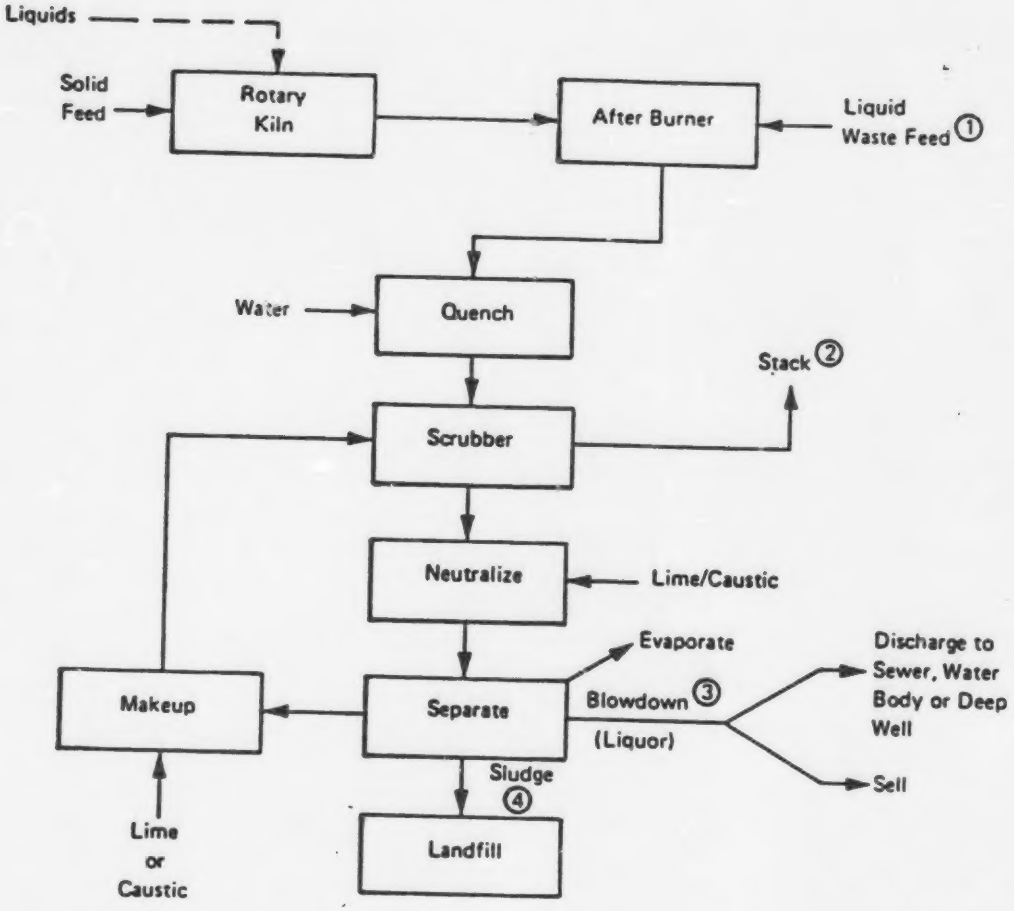

FIGURE F-1 PROCESS DIAGRAM

The scrubbed exhaust gases are released to atmosphere through the stack (Point 2). The gases contain the residual unburned wastes, products of incomplete combustion (PIC's), and inorganic ash. The scrubbing liquids may be treated by a variety of steps. The ones shown on the diagram are typical.

The liquids (also called liquor) are neutralized next for several reasons. The pH, which falls as the scrubbing solution contacts the acidic exhaust gases and absorbs hydrogen chloride, may be outside the permissible discharge limits and in that case will have to be adjusted by neutralization prior to discharge. Secondly, increasing the pH (more basic) will cause many of the heavy metal species, which are soluble in acidic media, to precipitate out of solution.

Following neutralization, the solids are separated from the liquor. The solids consist of particulate matter scrubbed out of the exhaust, and precipitated metals. Separation may be accomplished by filtration, centrifugation, or by settling in an impoundment. The latter two methods are used currently. An impoundment may also provide overall volume reduction through evaporation.

Some of the liquor that is separated from the solids must be discharged or purged from the system to prevent the chlorides which are scrubbed out from building up indefinitely (Point 3). This quantity of liquor, conventionally called blowdown, is adjusted so that the volume discharged contains an amount of chloride equal to that scrubbed out. Thus, at steady state, all the chloride leaves in the blowdown. Small quantities (insignificant, relative to blowdown) are carried out with the solids. For this analysis, the chloride concentration in the blowdown is assumed to be 0.7 percent. This figure is a composite derived from ranges found at existing facilities.

The blowdown is disposed of through different modes in the several major commercial incinerators operating currently (SCA, Chicago; ENSCO, El Dorado, Arkansas; Rollins, Deer Park, Texas, and Bridgeport, New Jersey). In one location the blowdown, which is in essence a concentrated brine, is sold in part as a feedstock. No further off-site discharge occurs which implies that the remaining brine is either disposed of in a deep well or is left to evaporate in an impoundment leaving the salts as crystalline solids. In other locations the blowdown is discharged indirectly through a municipal treatment plant or directly to a receiving water body.

These discharges must meet effluent limitations set by local authorities, so as to maintain water quality at or above the level prescribed in the water quality criteria. Chloride does not pose a risk when the permit conditions are met. Some metals have thresholds corresponding to no effect levels. For these, meeting permit limitations should result in negligible environmental or health effects. Other metals potentially are carcinogenic and may produce some risk at non-zero concentrations. Thus, there may be some risk to the exposed populations even if the discharges comply with permit limitations. The risk depends on the level used in establishing the standards for the receiving body.

The solids are disposed of in a secure landfill (Point 4). If an impoundment is used, the solids are collected periodically by dredging. The solids contain the particulate matter, metal precipitates and crystalline calcium or sodium compounds (depending on the alkali base used in the system).

The disposition of the heavy metals depends on the specific operations of the system. The metals entering in the waste are converted into solid particulate salts in the incinerator. These particulates are then removed in the scrubber. (Some small quantity will go up the stack as part of the uncollected particulates). Some of the solid materials may be (and generally are) solubilized by contacting the somewhat acid scrubbing solution. The part in solution, as noted above, may be precipitated, in whole or in part, in the neutralization step. Thus, in general, the metals are split into two streams; one in the liquor discharge and the other in the disposed solids.

To complete the process description, part of the liquor is recirculated to the scrubber after caustic or lime (or limestone) has been added to replenish the materials which have reacted with the gases.

Quantities Released

Two types of wastes have been considered, a PCB waste and a "typical" industrial waste. The PCB waste is assumed to contain 35 percent of Arochlor 1254 (53.6 percent chlorine) and 65 percent non-chlorinated hydrocarbons. The industrial waste is assumed to contain 50 percent ethylene dichloride (EDC, 73.2 percent chlorine) and 50 percent non-halogenated hydrocarbons.

Metals are present at the limits permitted by existing ocean-burning regulations.

The quantities of entering wastes, specific constituents in the wastes, and related quantities of the materials in the stack and the blowdown discharge are shown in Table F-1 and F-2 for the PCB and EDC wastes, respectively. The figures are expressed in metric tons per year. About 90 percent of the cadmium, chromium, and nickel and 50 percent of the arsenic are assumed to leave in the blowdown discharge. Also, the concentration of some of the metals in blowdown may be considerably less than the amounts shown, because the solubility in moderately alkaline media (pH 8 to 9) is of the order of 10-100 ppb.

Table F-1

QUANTITIES RELEASED: PCB WASTES
(In MT/YR)

Constituent	Feed (1)	Stack (2)	Blowdown (3)
Waste	56,000****	.056**	-
Chlorine	10,505 (18.9%)*	105	10,400 (0.7%)
Metals	22.4 (400 ppm)	4.5	17.9 (15.2 ppm)
Aqueous	-	-	1,486,000***

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* Values in ( ) are weight concentration
** 99.9999 percent DRE
*** For 300 Days/Yr = 1.3 million gallons/day (MGD)
**** 35 percent Arochlor 1254; rest non-halogenated hydrocarbons

Source: Arthur D. Little, Inc., as revised by IEc

Table F-2

QUANTITIES RELEASED: ETHYLENE DICHLORIDE (EDC) WASTES (In MT/YR)

Constituent	Feed (1)	Stack (2)	Blowdown (3)
Waste	68,400****	6.84**	-
Chlorine	25,034 (36.6%)*	250	24,784 (0.7%)
Metals	27.2 (400 ppm)	5.4	21.8 (7.6 ppm)
Aqueous	-	-	3,540,000***

---

* Values in ( ) are weight concentration
** 99.99 percent DRE
*** For 300 Days/Yr = 3.1 MGD
**** 50 percent EDC; rest non-halogenated hydrocarbons

Source: Arthur D. Little, Inc., as revised by IEc

Appendix G

ESTIMATION OF RISKS RESULTING FROM INHALATION OF HAZARDOUS EMISSIONS FROM OCEAN-BASED INCINERATORS

This appendix describes the estimation of human health risks from exposure to PCB and EDC emissions from an ocean-based incinerator operating in the Gulf of Mexico burn zone. The analysis described here addresses health risks resulting only from inhalation exposures. In this appendix, we present risk estimates for exposure to principal organic hazardous constituents (POHCs), products of incomplete combustions (PICs), and metals. Our description of the risk estimation methodology and the results from the analysis is divided into four sections:

1. Description of the atmospheric dispersion model,
2. Results from the dispersion model,
3. Description of the methodology used to convert the results from the dispersion analysis into human health risks, and
4. Analysis of the inhalation risk estimates.

Sections 1 and 2 of this appendix and Exhibits G-1 through G-22 were prepared by Applied Science Associates, Inc.

For estimation of human health risks from ocean-based incineration, the risk assessment approach described in this appendix relies on a variety of analytic techniques and computer models. Wherever possible, we have used practices developed or accepted by EPA. These models and techniques are not, however, perfect predictors of human health risks.

When using the results of our analysis, readers should be aware of the major assumptions and limitations underlying the approach. First, the risk assessment only considers risks from inhalation of pollutants by the the most exposed individual. These risk estimates are based on average annual air concentrations directly resulting from incinerator ship emissions. Long-term accumulation of pollutants in the atmosphere and the resulting risks are not considered. This may result in an underestimate of health risk. Second, field verification tests for the dispersion model used in our analysis have not been conducted for the study region. Calibrating the model for this site would result in more accurate estimates of air concentrations.

Finally, the PIC emission rates used to estimate human health risks are only very rough approximations of the quantities of PICs which might be present in the stack gases. This is particularly true for the dioxin and furan PICs generated in the PCB burns. One researcher with experience aboard the Vulcanus stressed the difficulty of knowing with any certainty the volumes of dioxin and furan PICs which might be generated in a burn. He stated that generation of small quantities of these substances appears to be a somewhat random process (personal communication, Dr. Donald Ackerman, Radian Corporation, Research Triangle Park, NC).

Because of the added uncertainty associated with the estimates of PIC emission rates for the PCB wastes, we include in this appendix an analysis of the sensitivity of the health risk results to alternative PIC emission rates. The alternative rate used is the PIC emission rate from land-based incinerators which is approximately 1000 times higher than the rate based on tests aboard the Vulcanus.

In general, the analysis of ocean-based incinerator risks is subject to the same uncertainties as the land-based incinerator analysis described in Appendix D. The results from our analysis suggest that the human health effects from inhalation of ocean-based incinerator emissions are lower than those for land-based facilities for both waste streams. The primary source of risk from both ocean- and land-based incineration is metal emissions. For the ocean-based alternative, most exposed individual risks due to metals is at least ten times lower than the risk at the land-based facilities described in Appendix D. For the emissions of POHCs, maximum individual risks are at least two orders of magnitude lower for the ocean-based facilities. For exposures to PICs from the PCB wastes, risks from at-sea incineration are substantially lower than those for land-based incineration, approximately six orders of magnitude. Even if the PIC emission rate for the land-based facilities is used in the analysis of ocean incinerator risks, PIC risks from the ocean option are still lower by approximately three orders of magnitude. PIC exposures for the EDC waste are also roughly ten times lower for the ocean-based system.

ATMOSPHERIC TRANSPORT MODEL DESCRIPTION

An atmospheric transport model has been used to estimate on-shore levels of exposure to PCB and metal stack emissions from the Gulf of Mexico ocean incineration site. The model used here is based on the particle-in-cell, Eulerian-Lagrangian approach to solution of the mass transport equation (Lange 1973). A 0.25 degree spherical grid system is used for transport computations, whereas concentration levels are computed on a 0.09 degree grid. This smaller grid size is dictated by the typical plume width at the coastline. The coastal outline of the transport grid system is shown in Exhibit G-1. Although the model is three-dimensional in formulation, wind data limitations dictate a two-dimensional application in the present case.

An annual cycle of winds is used to drive the model. This cycle is a synthesized climatological characterization of the boundary layer winds used originally to drive a Gulf of Mexico ocean circulation model (Blumberg and Mellor 1979). The original data (TDF-11) are maintained by the National Climatic Center, and consist of over one million surface ship observations in the Gulf of Mexico. These data were statistically reduced to obtain monthly mean wind fields. Diurnal and four-day (storm) frequency information was then re-introduced (Blumberg and Mellor 1981). Here we have removed the diurnal variations, on the assumption that horizontal dispersion coefficients adequately reflect turbulence associated with frequencies greater than once per day.

In the western Gulf of Mexico, the daytime atmospheric boundary layer depth varies from 1 km (winter) to 2 km (summer). The nighttime depth is typically smaller by a factor of 2. In extreme cases the nighttime depth can be as low as 300 m, and in a "worst case" situation this value can persist for several days. The mixed layer depth estimates used here are climatological values based on average of radiosonde soundings at coastal stations. These are the results of Heffter (1980) and take into account the existence of critical temperature inversions in the soundings used to estimate the layer depth.

The near field (momentum and bouyancy dominated) plume dynamics are parameterized by using an effective release height instead of the actual stack height above sea level. This effective release height is computed as a function of the gas exit temperature and velocity, the stack diameter, and a cross-stack wind speed (Blevins 1984). For gas exit temperature of

1400 degrees K, exit velocity of 15 m/sec, and stack diameter 3.4 m (ICAIR 1984), final plume rise height will be between 70 and 300 m, depending on weather conditions. A value of 150 m has been used as a typical year-round value for these simulations. This is less than the 325 m height assumed by Weitkamp et al. (1984) over the North Sea, but corresponds well to estimates by Schatzmann and Policastro (1981).

The vertical diffusivity Kz is computed as a function of the friction velocity uf, the altitude z, Coriolis parameter f, the von Karman constant k, and a stability factor c (Schayes and Cravatte 1980):

$$K_z = z k u_f \exp(-c \ (zf/ku_f)^{0.764}) \qquad (2\text{-}1)$$

The friction velocity is:

$$u_f = kU/(\log \ (z/z_o) \qquad (2\text{-}2)$$

in which U is wind speed at the reference height z (= 10 meters here), and zo, the roughness length, is estimated from Wu (1984) as

$$z_0 = 0.0533 \exp(-0.837 \ U) \qquad (2\text{-}3)$$

for U in m/sec. The parameter c varies with stability conditions as shown in Exhibit G-2.

Dry deposition velocities (Vs, in m/sec) of particulates are computed according to Stoke's law:

$$V_s = 3.0 \times 10^{-5} D^2 S \qquad (2\text{-}4)$$

for D = particulate diameter in microns and S = specific gravity of particulates. The constant in Equation 2-4 is applicable for standard atmosphere near 20 degrees Celsius (Raridon et al. draft). All gases are assigned a settling velocity of 0.001

m/sec, and gases contacting the ocean surface are not returned to the atmosphere (absorbing boundary). The top of the atmospheric boundary layer is treated as a reflecting boundary for both particulates and gases.

Wet deposition, or washout due to rainfall, is modeled as a stochastic process based on long term synoptic weather observations (U.S. Dept. Commerce). We use a first order process:

$$dQ/dt = -hQ \tag{2-5}$$

where Q is the amount of material in a column of atmosphere, and h (per second) is given by:

$$h = 5.55 \times 10^{-4} R^{0.6} K_d \tag{2-6}$$

(Slade 1968). Here R is rainfall rate (mm/hr) and Kd is molecular diffusivity (cm2/sec), which can be estimated from

$$K_d = 1.09/ M^{0.5} \tag{2-7}$$

in which M is the molecular weight of the pollutant (Lyman et al. 1982). Equation 2-7 is valid near 20 degrees C and 1 atm. pressure. The rainfall rate R is drawn from a probability distribution based on Exhibit G-3. Frequencies are derived from 10 years of historical data for Brownsville, Texas. Monthly average rainfall amounts are long term (> 50 yr.) values. Rain storms are assumed to occur synoptically over the entire plume. Model input parameters are summarized in Exhibit G-4.

PLUME TRAJECTORY AND ATMOSPHERIC CONCENTRATION ESTIMATES FOR ARBITRARY SOURCE STRENGTHS

The burn scenario and resulting atmospheric concentrations simulated here are based on a total incineration ship waste load of 4,000 metric tons (mt), and an incinerator throughput of 15 mt/hr. These data imply an 11 day burning operation which, when coupled to an assumed 15 day interval for travel and load time, gives a 26 day cycling time for the full operation. This 11 day burn time is less than the 12.3 day maximum, and greater than the 6 day minimum time estimated in Appendix A. To give a complete picture of the annual wind cycle, we have simply modeled an 11 day burn beginning on the first of each month.

Exhibits G-5 through G-16 show (a) the mean wind field for the fist 12 days of each month, and (b) stack emission concentration contours at the end of days 1, 2, 11, and 12 in each burn cycle.

The annual variability in the predominant wind direction is evident in the shifting location of the incineration plume (Exhibit G-5b through G-16b). February is at one end of the annual cycle, when the plume follows its most southwesterly track. As the winds shift in the spring the plume gradually moves northward along the coast, although maintaining its westward orientation. In July and August, at the other end of the cycle, the winds carry the incineration plume almost due north. The fall and winter months reverse the trend of the spring as the westward winds shift towards a southern orientation.

POHCs and PICs

Exhibits G-17a through G-17l show maximum PCB concentration levels at the coastline as a function of time for each burn cycle assuming a stack emission rate of 1 kg/hr.1/ A typical rural background concentration of 1 ng/m3 (U.S. E.P.A., 1981) is shown on each plot, and the increased PCB concentration due to incineration is added to this ambient level. (For urban areas, with typical backgrounds of 5-10 ng/m3, the contribution of the incineration plume would be a proportionately smaller fraction of the ambient concentration.) These time series (Exhibit 17) are Lagrangian in the sense that they follow the plume as it wafts across the coastline, and reflect the maximum concentration

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1/Although the discussion of POHC transport in the plume is framed in terms of PCBs, the reader should note that because both substances are emitted as gases, they are likely to have similar settling velocities. Hence we use these modeling results for both PCBs and EDC. The PCB dispersion analysis is also assumed to be appropriate for estimating exposures to PICs.

within the plume. These figures are useful in demonstrating the degree of predicted concentration variability due to wind field variations as well as rainfall events. Rain occurs as a stochastic process in the model, and the number of events occurring during each simulated burn cycle is shown in Exhibit G-18. Seasonal probabilities and mean intensities are taken from Exhibit G-3. Each rain event is assumed to have a duration of 24 hours.

In general, the summer months (April through September) result in lower maximum coastal exposure levels than the winter months. This fact is due primarily to the doubling of the mean atmospheric boundary layer during the summer (Exhibit G-4). An exception to this pattern is the December simulation, during which 6 rain days occurred (by random probability) during the 11 day burn.

We assumed that PCB emitted from the stack is 90% gaseous, 10% particulate, or rapidly conforms to the observed background particulate distribution (EPA, 1981). The gaseous deposition rate has been estimated to lie between 0.05 and 1 cm/sec (EPA, 1981). Exhibits 17a-171 are based on a deposition velocity of 0.1 cm/sec (Exhibit G-4). For comparison, Exhibits G-19a and G-19b show the January time series for deposition velocities of 0.2 and 0.5 cm/sec. The deposition velocity is clearly an important model parameter, and it should be noted that our choice of a low value may result in overestimation of both maximum concentrations and integrated exposure levels.

In order to estimate worst case exposure levels, it is necessary to evaluate concentration time series at fixed coastal locations. The maximum fixed location (Eulerian) exposure level has been found in the following way. First it is clear that the winter months result in higher coastal concentration than the summer months (Exhibit 17) due to the decreased boundary layer thickness. A north-south column of 85 grid cells approximately coincident with the west coast of the Gulf of Mexico was therefore selected for further evaluation. A program was then written to compute the time-integrated concentration level in each of these grid cells, and the coordinates of the maximum value were recorded. The non-zero monthly time series at this location, approximately at 24.5 north latitude and 97.8 west longitude, near Brownsville, Texas, are shown in Exhibits 20a-20c. The net annual PCB exposure for an individual living in this area is computed to be about 11 ng-days/m3. This is the maximum modeled value for any location along the coastline. The maximum peak concentration, occurring also at this site in the model record, was about 6 ng/m3, and lasted for about 12 hours.

Metals

The following analysis is based on a generic metal with density, characteristic particle size, and molecular weight equal to the mean values for arsenic, cadmium, chromium, and nickel (Exhibit G-4). This metal has a dry deposition velocity of 0.08 cm/sec, or nearly the same as the gaseous dry deposition rate, 0.1 cm/sec, which controls 90% of the PCB emissions. Because deposition occurs at roughly the same rate for both gases and metals, differences in their distributions in the physical environment are due primarily to differences in their mass rate of emission from the incinerator stack.

The governing winds and path of the plume from the stack for each month of the year are shown in Exhibits G-5b through G-16b, although the concentration values noted in those figures refer to PCB released at 1 kg/hr. We assume that the metal is present in the waste stream at 100 ppm. For a feed rate of 15 mt/hr, the release rate for our generic metal is 1.5 kg/hr, and because the deposition rates are similar, the PCB concentration values of Exhibits G-5b through G-16b can be multiplied by 1.5 (for each metal) to give good estimates of the accompanying metal concentrations. This comparison is only approximate since 10% of the PCBs emitted are assumed to be in particulate form, with a fall velocity of 0.45 cm/sec, and because the molecular weight of PCB (as 1254) is more than 4 times that of the metal, resulting in a slower rainout state.

Maximum modeled metal concentrations in the plume at the coast are shown in Exhibits 21a through 21d for February, May, August, and November. Comparison with the corresponding PCB coastal concentrations (Exhibits 17b, e, h, and k) shows that the 1.5 multiplier for the mean concentration level is only approximate. Furthermore, the metal concentrations were simulated separately form the PCB model runs, so that stochastic rain events do not occur at the same time. It is therefore not meaningful to compare specific peak values between Exhibits 17 and 21.

Maximum fixed point (Eulerian) exposure levels for metals occur at the same location as for PCBs. Exhibit G-22 shows the concentration time series at this location for the November burn, the highest coastal exposure month modeled. Exhibit G-22 is the fixed location time series corresponding to the maximum concentration (moving location) time series of Exhibit G-21d. The net annual exposure at this point is 17 ng - day/m3 (11.3 ng-day/m3 if one assumes emissions of one kilogram per hour).

RISK ESTIMATION METHODOLOGY

To estimate health risks for the most exposed individual, we employed a four step process:

1. First, we determined the actual emissions of POHCs, PICs and metals from the incinerator ship;
2. Second, based on the results from the 1 kilogram per hour dispersion modeling results, we estimated the dose which the most exposed individual would receive over a year.
3. Third, we scaled this dose to reflect the dose which would have resulted from the actual level of pollutant emissions from the ship; and
4. Fourth, we converted this maximum dose to an estimate of cancer risk.

These steps were performed separately for the PCB and EDC wastes. As noted earlier, we also assessed the sensitivity of the results to an alternative assumption about PIC emission rates for the PCB waste. Each of these steps is discussed below in greater detail.

In estimating the POHC emission rate when the ship is burning PCBs, we have assumed a waste feed rate of 15,000 kilograms per hour. Thirty-five percent of the waste is comprised of PCBs (5,250 kg/hr), and we assume the remainder is composed of non-hazardous materials. Assuming that the ship achieves a destruction and removal efficiency (DRE) of 99.9999 percent, PCB emissions will be 5.25E-3 kilograms per hour. For the EDC waste stream, we assume a waste feed rate of 25,000 kilograms per hour of which 12,500 kilograms is assumed to be non-hazardous. Based on the regulatory requirement of 99.99 percent DRE, EDC emissions are 1.25 kilograms per hour.

Estimates of emission rates for PICs were based on data from incinerator trial burns. For the PCB waste, we initially set PIC emissions equal to 1.11E-11 percent of the PCB waste feed rate. Although no TCDDs were detected in the analysis of stack gases from PCB burns aboard the Vulcanus, this value was selected because at quantities below it, the sampling methods used would not have detected TCDDs even if they had been present (see Appendix E). For the analysis of the sensitivity of the results to alternative PIC emission rates, the rate for the land-based incinerator burning PCB wastes was employed (3.40E-8 percent of the waste feed rate). For the EDC waste, PICs were assumed to be 3.01E-2 percent of the EDC feed rate. A more detailed description of the PIC estimation methodology appears in Appendix E.

To estimate emission rates for metals, we assumed that 10 kilograms per hour of carcinogenic metal compounds are fed to the incinerator in the EDC waste. For PCB wastes, we assume 6 kg/hr of carcinogenic metals. Since metals are not destroyed by the incineration process, all metals fed to the facility are emitted in the stack gases.

The second step in the risk estimation process is the determination of the maximum average daily dose which a person living along the Gulf Coast would receive each year as a result of incinerator ship stack emissions. This was done by identifying the coastal grid point with the highest annual exposure, where annual exposure is defined as the ambient on concentration integrated over the entire year. Coastal grid points were searched to identify the one with the highest integrated exposure. This occurs near Brownsville, Texas. The integrated exposures for PICs, POHCs and metals at this location are shown in Exhibit G-23. The integrated exposure for metals is very similar to the value for PICs and POHCs. As noted earlier, this occurs because the deposition velocities for these metals and gases are roughly equivalent. Approximately 60 percent of each compound is deposited to the surface of the ocean before the plume reaches the shoreline.

To convert the integrated exposure (ng-days/m3) to an estimate of individual dose (mg/kg/day), we performed the conversion illustrated in Exhibit G-24. In this calculation the integrated exposure is multiplied by factors accounting for the average volume of air breathed per day (22.8m3), the average adult bodyweight (70kg), and the proportion of the pollutant retained in the body (0.65). In addition, the units are converted from nanograms to milligrams. This calculation yields the total pollutant dose during the year (mg/kg/yr). This is then divided by 365 to get the average daily dose (mg/kg/day). This dose represents the highest dose any individual living along the Gulf Coast would receive due to pollutants emitted at a rate of one kilogram per hour from the incinerator ship.

The maximum dose estimates based on the one kilogram per hour emission rate for PICs, POHCs, and metals are presented in Exhibit G-23. For PICs and POHCs, the dose is 6.38E-9 mg/kg/day. For metals, the most exposed person receives a dose of 6.55E-9 mg/kg/day.

The third step in the risk estimation process is to scale this maximum average daily dose for one kilogram per hour of emissions to reflect actual emissions from the incinerator. Since doses are a linear function of the source emission rate, this scaling can be accomplished by multiplying the maximum dose (6.38E-9 for PICs or POHCs and 6.55E-9 for metals) by the ratio of actual to modeled emission rates. The actual emission rates used in this analysis are included in Exhibit G-25. The calculation of average daily doses is illustrated in Exhibit G-26. This calculation is performed for PICs, POHCs, and metals for both the PCB and EDC wastes.

The resulting estimates of maximum average daily doses (mg/kg/day) are presented in Exhibit G-27. These range from 1.08E-17 for the PCB POHCs to 6.55E-8 for the metals in the EDC waste. In the PCB case metals account for 99.9 percent of the dose. In the EDC case, metals account for only 54.1 percent of the total dose. This difference is explained by the lower DRE and higher PIC emission rate assumed in the EDC analysis. This results in higher PIC and POHC emissions relative to the metals.

In Exhibit G-28, we compare the maximum average daily doses assuming the PIC emission rate for land-based incinerators burning PCB wastes with those based on the PIC emission rate for ocean-based incinerators. Even in the case where we assume a PIC generation rate 1000 times higher than the one actually calculated for the incinerator ship, the PIC dose to the most exposed individual only increases to 13.25E-14.

The final step in the process is the estimation of risks. To estimate risk for the most exposed individual, the average daily dose of a chemical is multiplied by its cancer unit risk factor. The unit risk factor (1/mg/kg/day) is a measure of the risk generated by a dose of 1 milligram per kilogram of bodyweight per day received continuously over a 70 year period. This produces an estimate of the lifetime cancer risk borne by a person living at the maximum exposure location and breathing the pollutant continuously over 70 years.

The cancer unit risk factors used in this analysis are presented in Exhibit G-29. For POHC emissions, we used the unit risks for the primary pollutant, either PCB or EDC. For PICs, however, the determination of the appropriate unit risk factor was complicated by the large number of compounds appearing as PICs in the stack gases. For the PCB waste streams, only the PICs appearing in the trial burns for which data were available were dioxins and furans. In estimating the risk from exposures to mixtures of dioxins and furans, we followed EPA recommendations by applying the unit risk factor for 2,3,7,8-TCDD to the dose of TCDD PICs. (Personal communication with Dr. Donald Barnes, EPA Office of Toxic Substances, September 1984.) The unit risk factor used for the metals is a weighted average of the risk factors for the four carcinogenic metals emitted in the stack gases (arsenic, cadmium, chromium, and nickel).

For the EDC waste stream, we selected chloroform as the representative PIC. Although this compound did not appear in the analyses of the three burns aboard the Vulcanus for which we have data on PICs, it appeared in five of the eight trial burns of non-PCB organic wastes which took place at land-based units and represented on average 59 percent of the total mass of PICs in the sampled stack gases (see Appendix E). The unit risk factor for chloroform is approximately the median value for the carcinogens appearing in the analyses of these trial burns. The unit risk factor for the metals is again the weighted average risk factor for the four carcinogenic metals in the waste.

ANALYSIS OF INHALATION RISK ESTIMATES

Exhibit G-30 summarizes the human health risks due to inhalation of hazardous pollutants emitted from ocean-based incinerators. The resulting risk estimates are for the most exposed individual. The analysis suggests that maximum individual risks are relatively insignificant for both the PCB and EDC wastes. Total risk to the most exposed individual for the PCB wastes is 6.37E-7, roughly 6 chances in 10,000,000 over a

70-year lifetime. For the EDC wastes, the risk is approximately 1 chance in 1,000,000 over 70 years.

For the PCB waste, more than 99.9 percent of the risk results from inhalation of metals. Inhalation of POHCs and PICs represent an insignificant proportion of the total risk. For the EDC waste, 99.6 percent of the risk derives from inhalation of metals. Both PIC and POHC emissions associated with the EDC wastes result in trivial risks. Since other persons living in the area affected by the plume would receive doses less than or equal to the dose on which this MEI risk is based, the most exposed individual risks represent upper bounds on individual risk.

Although we have not calculated an explicit estimate of total cases resulting from incinerator emissions, the value will be quite low. The average risk to people living in the Gulf Coast region will be considerably less than the maximum risk of 1.06E-6. As a result, per million people exposed over a 70 year period, substantially less than one case will result.

Exhibit G-31 compares the risks from ocean incineration with those resulting from the use of land-based incineration PIC generation rate. Risks from PICs under this alternative assumption increase from 1.68E-12 to 2.15E-9. Under the alternative assumption MEI risks due to PIC emissions are still only 3 in 1,000,000,000 well below the level which EPA has in the past considered cause for concern.

To place the human health risks from ocean incineration in perspective, we compare these risks to those resulting from exposure to other hazardous air pollutants. In the recently published EPA draft report titled The Magnitude of the Air Toxics Problem in the United States (EPA, 1984), available estimates of the risks from exposure to airborne toxics are summarized. This suggests that, relative to the risks from other significant air toxics to which people are exposed, maximum risks from inhalation of ocean-based incinerator stack emissions are relatively minor. For persons receiving lower doses than the most exposed individual, the incremental contribution of ocean-based incineration risk will be even less significant. The average individual exposed to air toxics in the U.S. faces a lifetime risk from these compounds of between 4 and 6 in 10,000 (Exhibit G-32). Comparing our estimates of risk to the most exposed individual resulting from ocean-based incineration, the incremental risk from incineration would increase the average risk from air toxics by less than 1 percent.

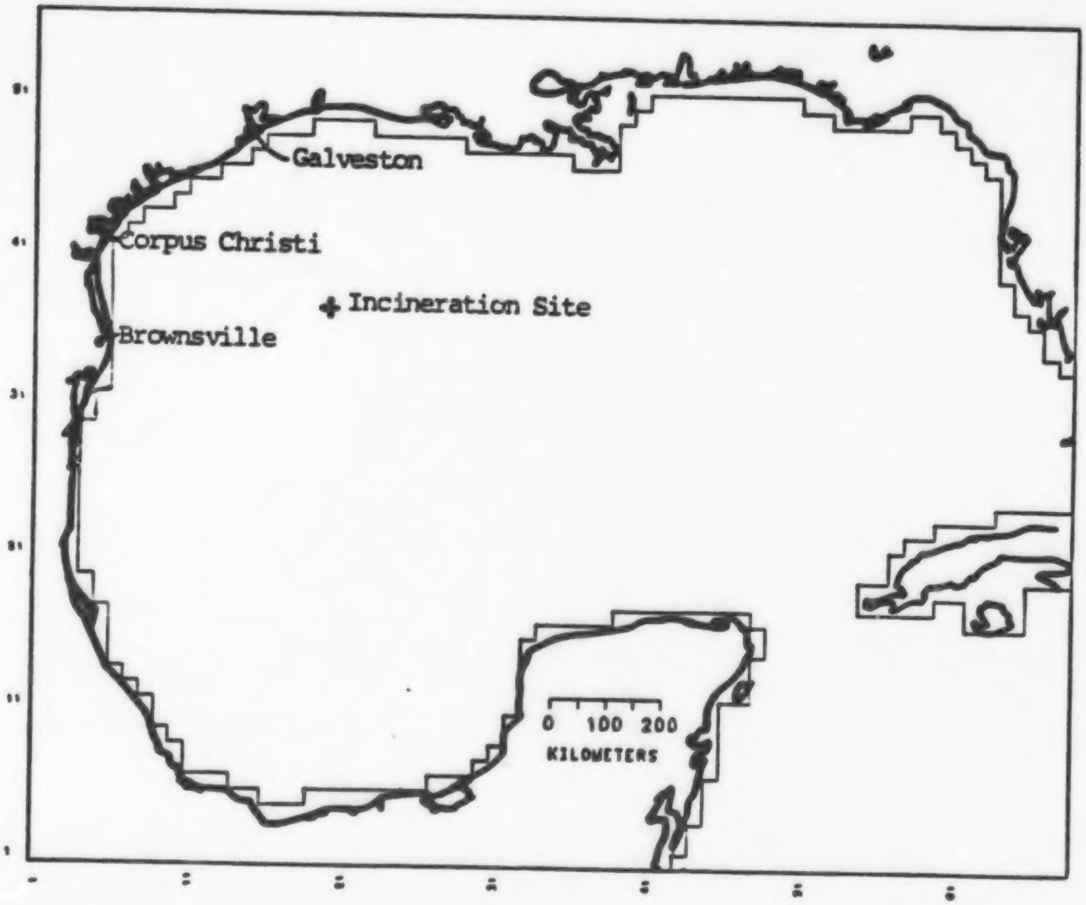


Exhibit G-1 Study area, showing model grid system coastal outline, coastline, incineration site.

Exhibit G-2. Values of the stability parameter c (from Shayes and Cravatte, 1980)

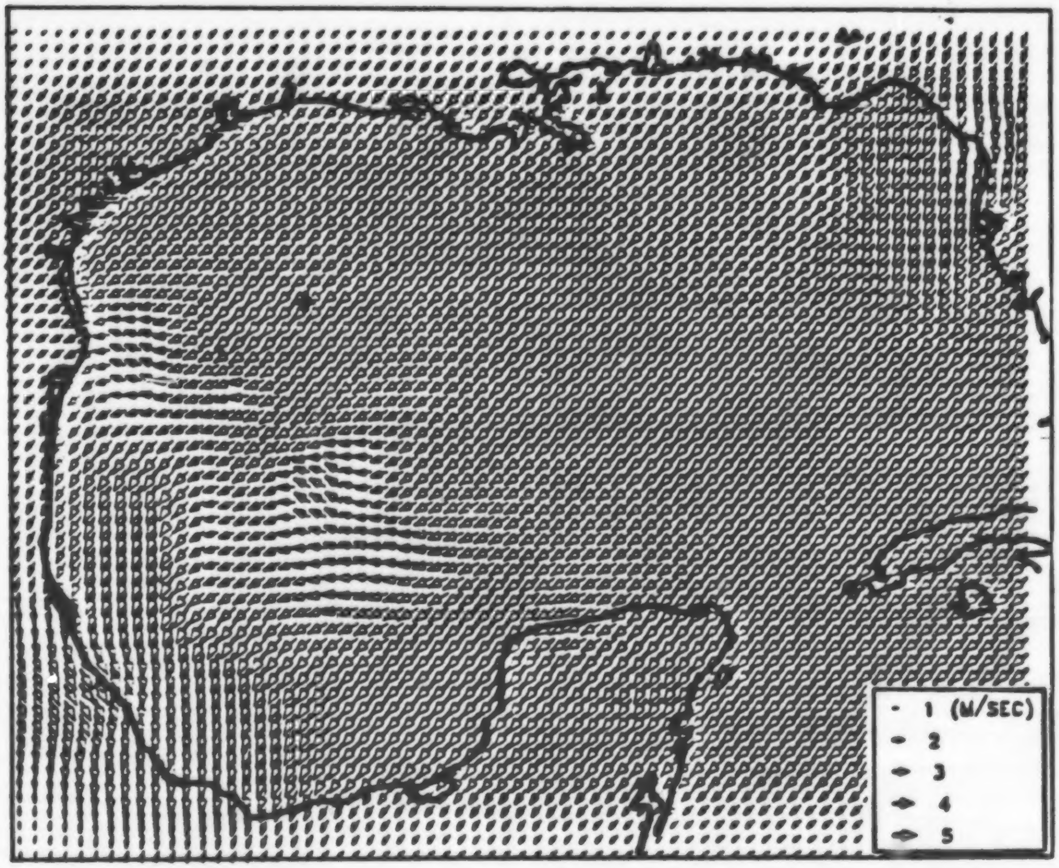
c	Stability
27.8	very stable
14.8	stable
7.8	neutral
3.0	unstable
1.6	very unstable

Exhibit G-3. Rainfall data summary for Brownsville, Texas. (a) U.S. Dept. Commerce, Climatic Data National Summary 1970-1980, (b) World Weather Records 1983

Season	Frequency (a) (days/mo.)	Amount (b) (mm/mo.)
Dec-Jan-Feb	7	39
Mar-Apr-May	4	42
Jun-Jul-Aug	7	63
Sep-Oct-Nov	8	84

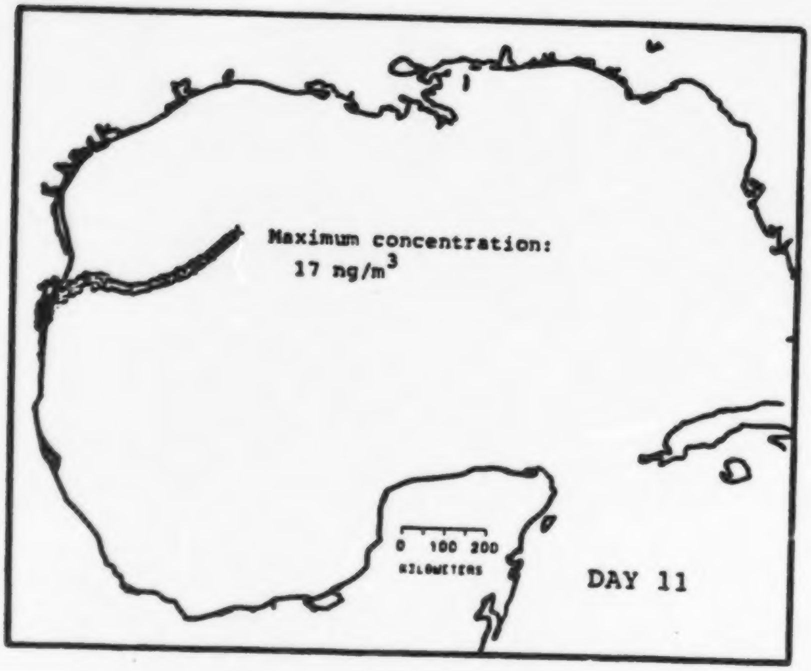
Exhibit G-4. Parameter values for atmospheric model applications.

Parameter Name	Symbol	Value
von Karman constant	k	0.4
stability coefficient	c	7.8
Coriolis parameter	f	7.3×10^{-5}
particle diameters	Di	(micrometers)
-arsenic		1
-cadmium		2
-chromium		3
-nickel		1.5
-generic metal		1.9
-PCB		10
particle densities	S_i	(gm/cm^3)
-arsenic		5.7
-cadmium		8.7
-chromium		7.1
-nickel		8.9
-generic metal		7.6
-PCB		1.5
molecular weights	M	(gm/mole)
-arsenic		75
-cadmium		112
-chromium		52
-nickel		59
-generic metal		75
-PCB		326
atmospheric boundary layer		
-summer		1.5 km
-winter		0.75 km
diffusivities	Kx,Ky	1000 m^2/sec
gaseous PCB deposition rate		0.1 cm/sec
gaseous/particulate PCB mass emission ratio		9



GULF OF MEXICO
AVERAGE WIND FIELD FOR FIRST 12 DAYS OF JANUARY

Exhibit G-5a Average wind field for the first 12 days of January.



Contour intervals: 0.1,1,5,10,15 ng/m^3

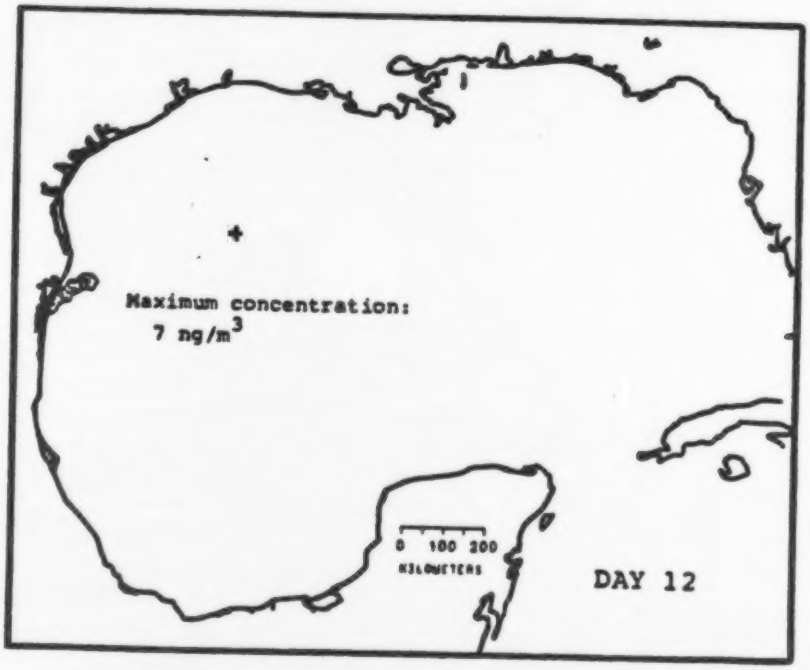
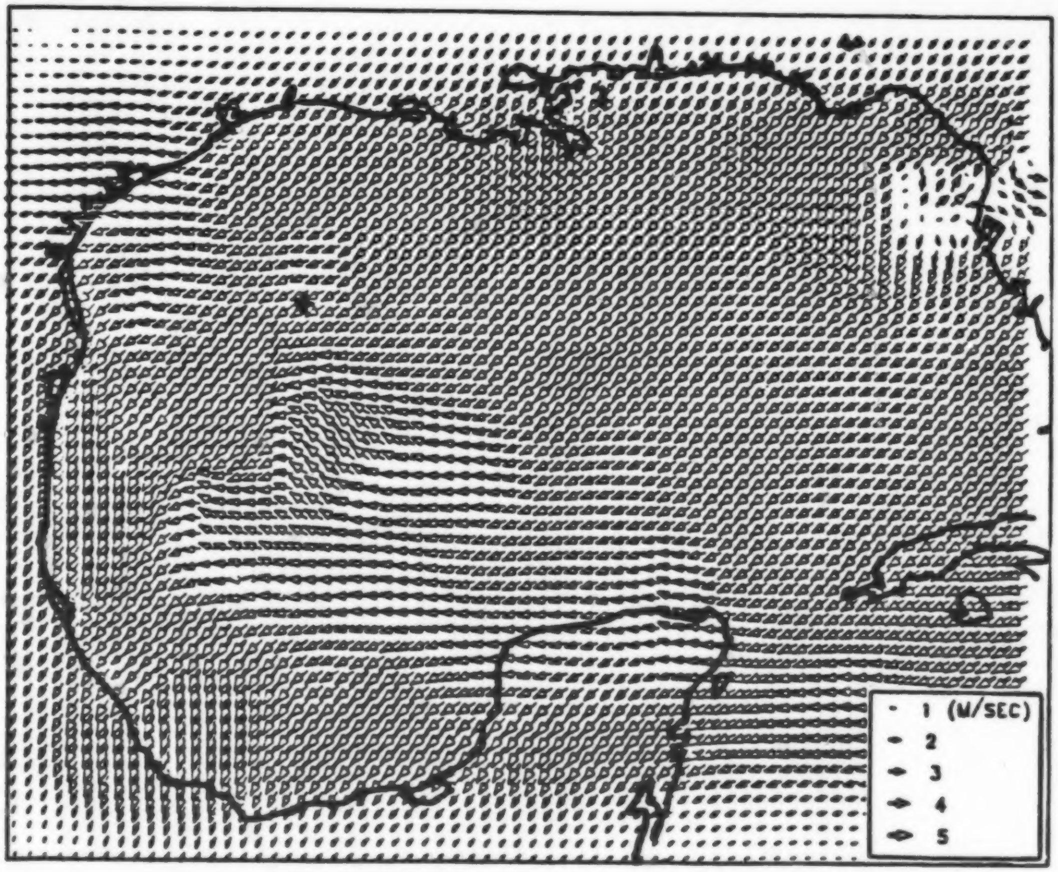
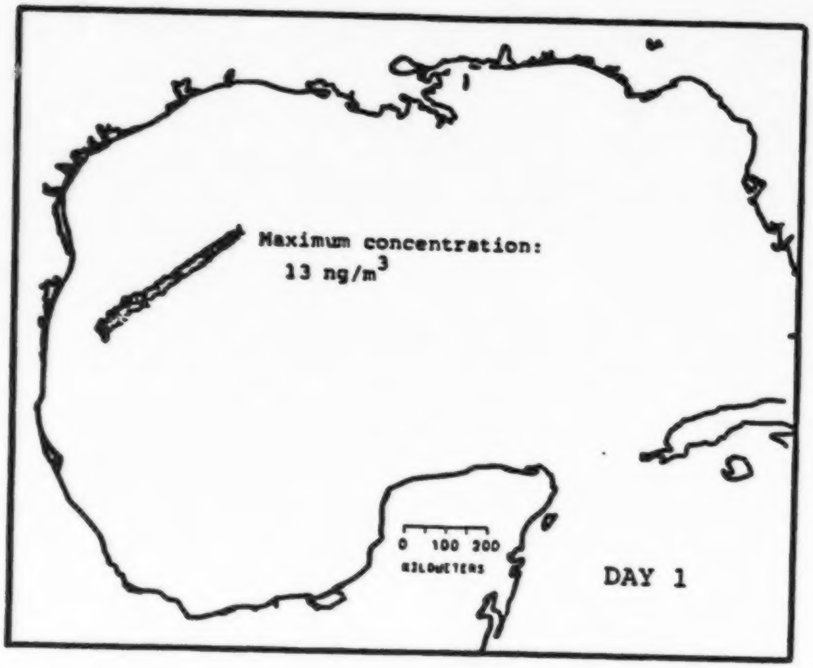


Exhibit G-5b cont. Incineration plume concentration contours during and after an 11 day burn beginning January 1. Background contaminant levels are neglected in computing concentration values.



GULF OF MEXICO
AVERAGE WIND FIELD FOR FIRST 12 DAYS OF FEBRUARY

Exhibit G-6a Average wind field for the first 12 days of February.



Contour intervals: 0.1,1,5,10,15 ng/m^3

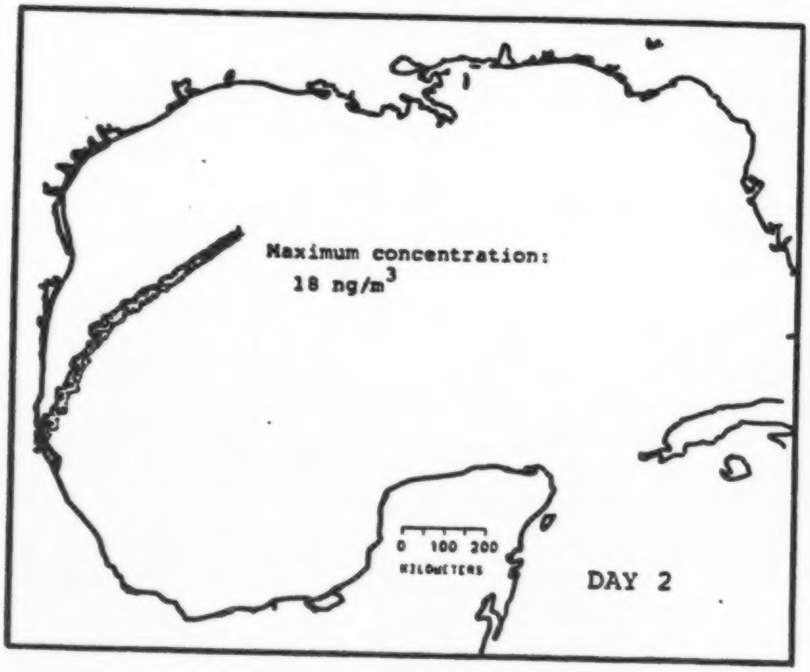
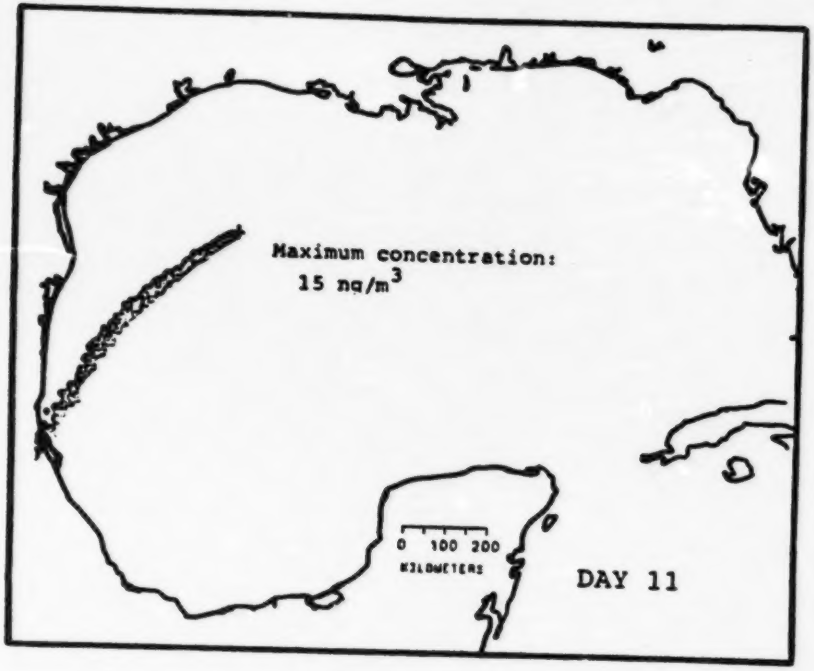


Exhibit G-6b Incineration plume concentration contours during and after an 11 day burn beginning February 1. Background contaminant levels are neglected in computing concentration values.



Contour intervals: 0.1,1,5,10,15 ng/m^3

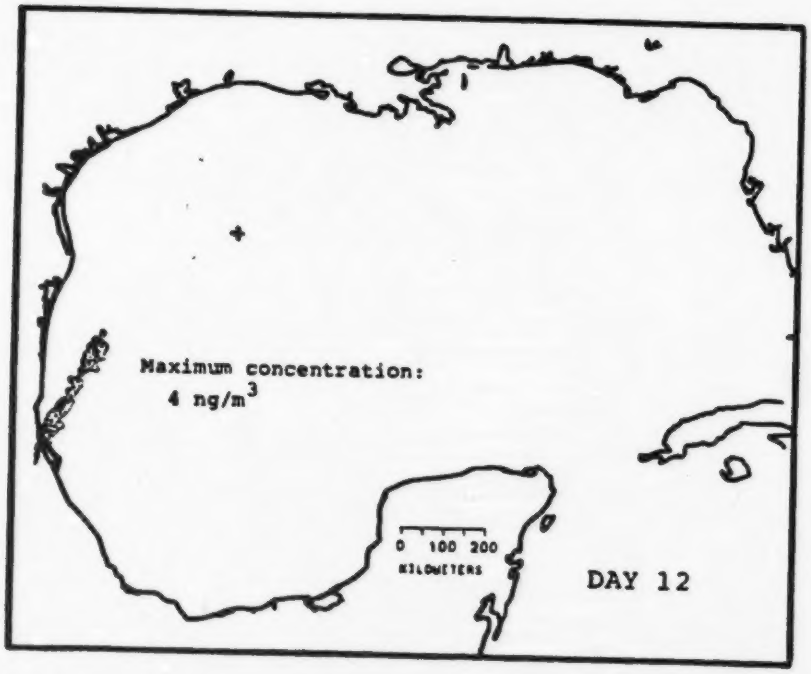
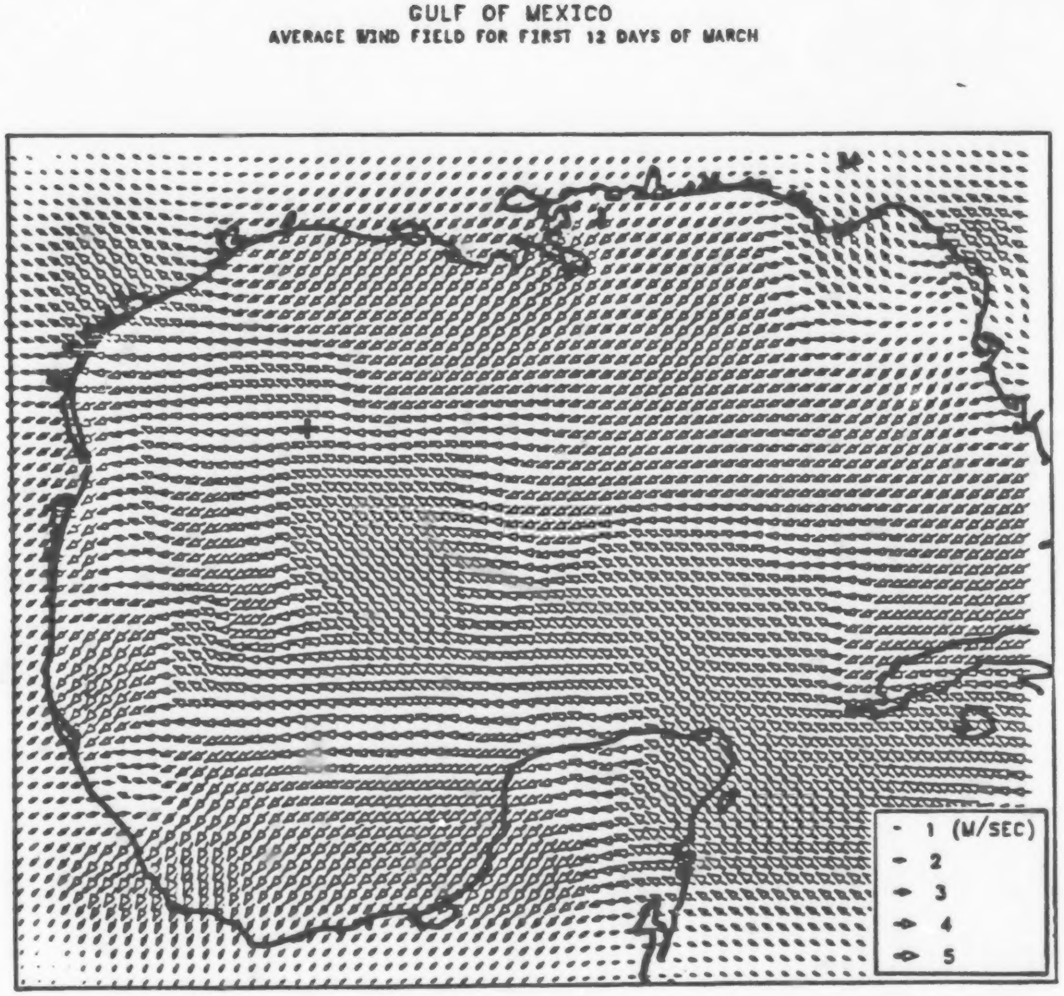
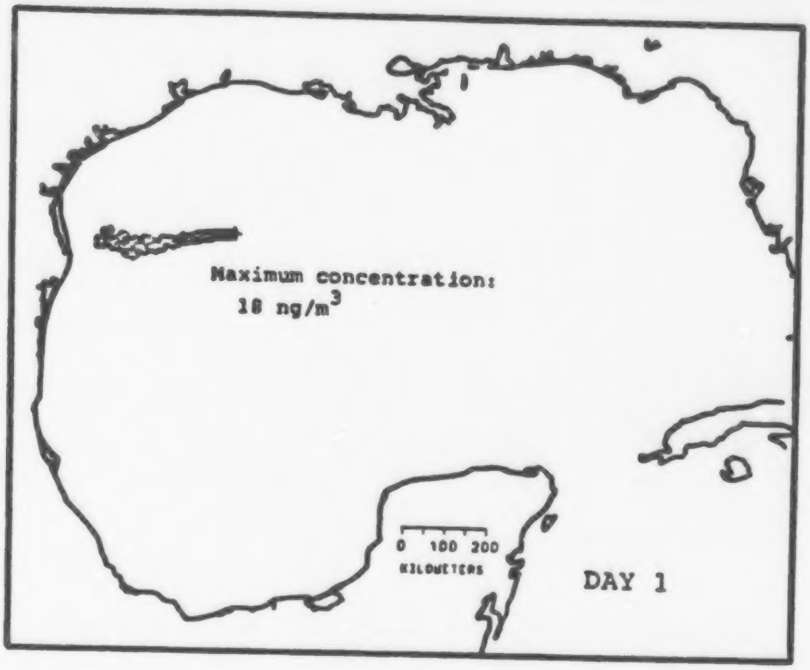


Exhibit G-6b cont. Incineration plume concentration contours during and after an 11 day burn beginning February 1. Background contaminant levels are neglected in computing concentration values.



GULF OF MEXICO
AVERAGE WIND FIELD FOR FIRST 12 DAYS OF MARCH

Exhibit G-7a Average wind field for the first 12 days of March.



Contour intervals: 0.1,1,5,10,15 ng/m^3

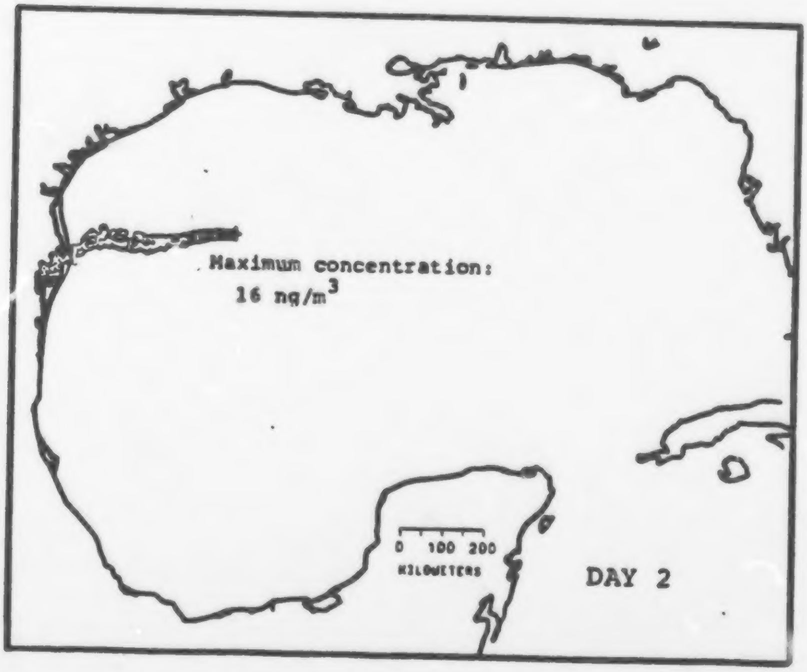
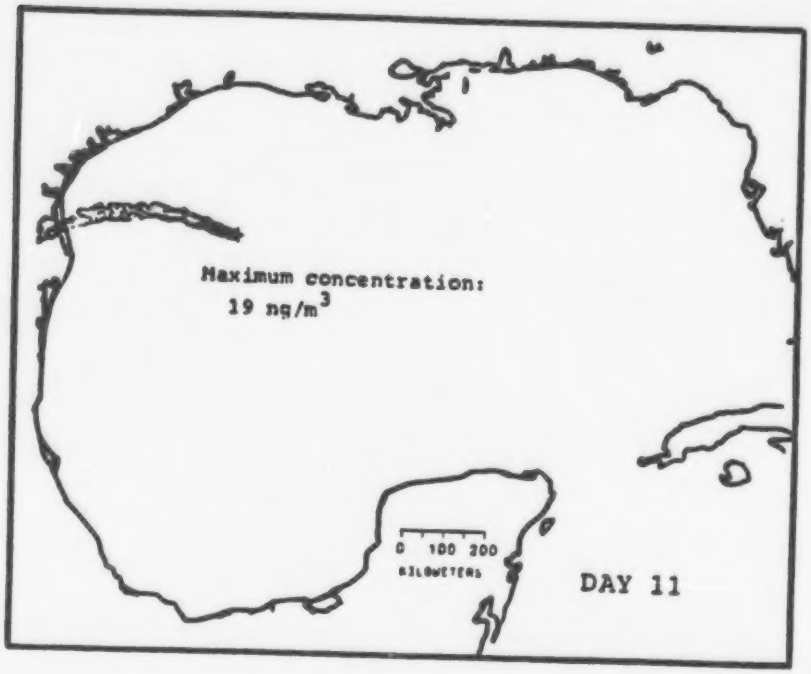


Exhibit G-7b Incineration plume concentration contours during and after an 11 day burn beginning March 1. Background contaminant levels are neglected in computing concentration values.



Contour intervals: 0.1,1,5,10,15 ng/m^3

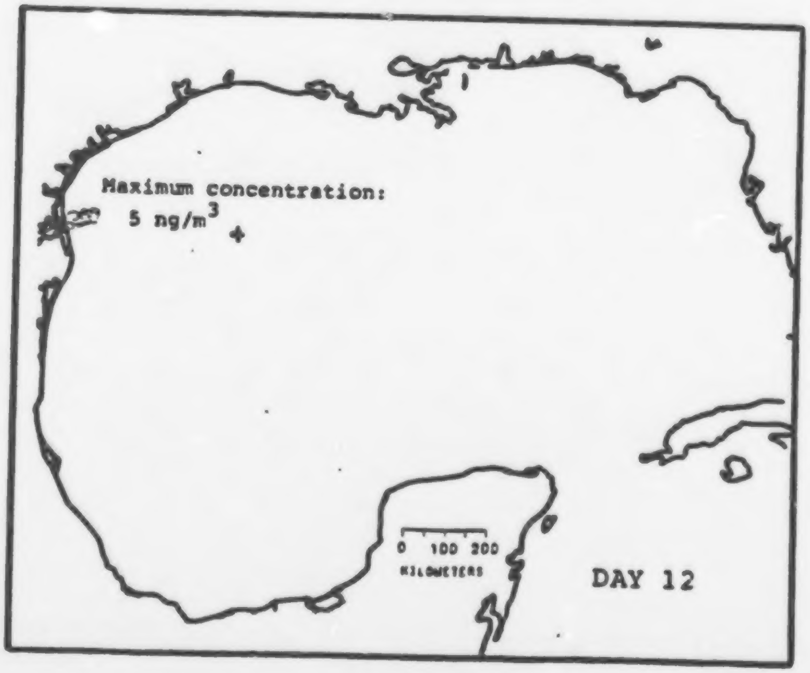


Exhibit G-7b cont. Incineration plume concentration contours during and after an 11 day burn beginning March 1. Background contaminant levels are neglected in computing concentration values.

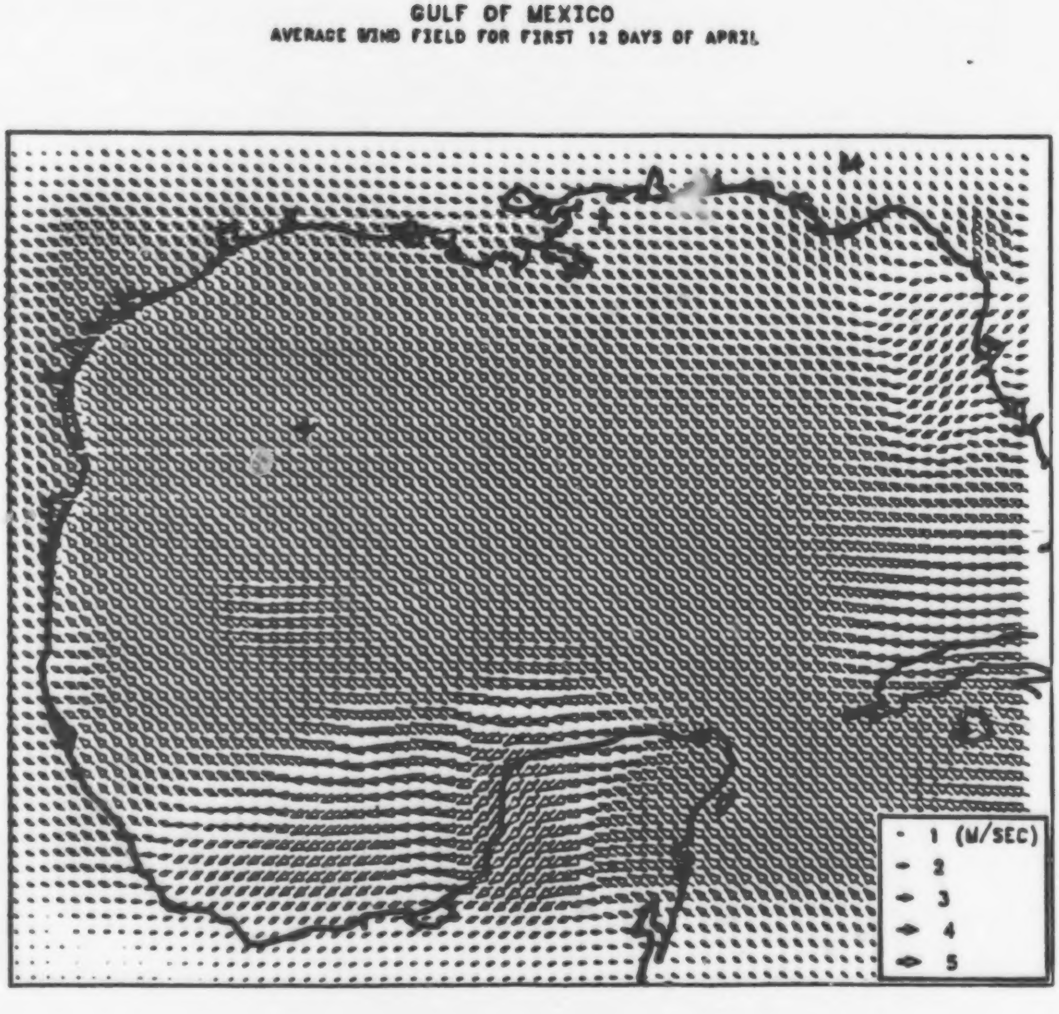
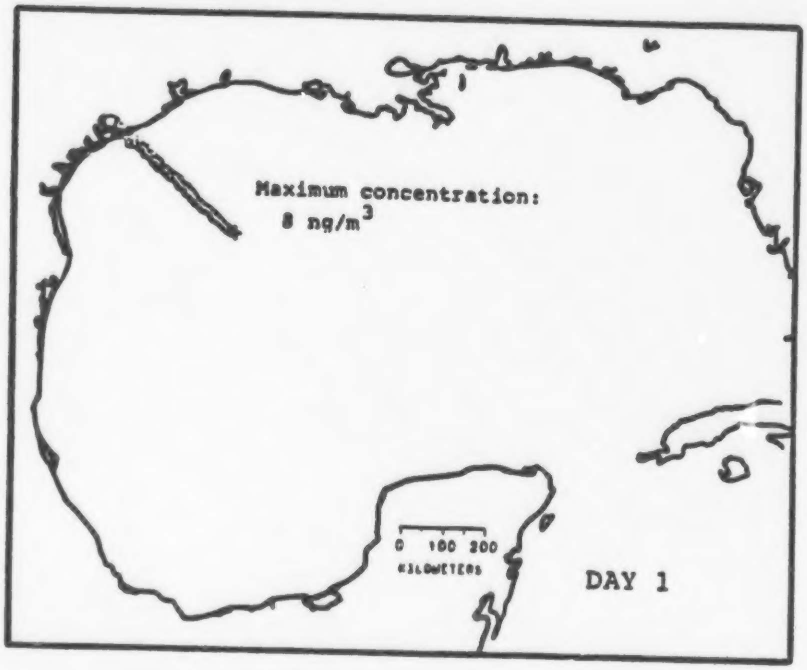


Exhibit G-8a Average wind field for the first 12 days of April.



Contour intervals: 0.1,1,5,10,15 ng/m^3

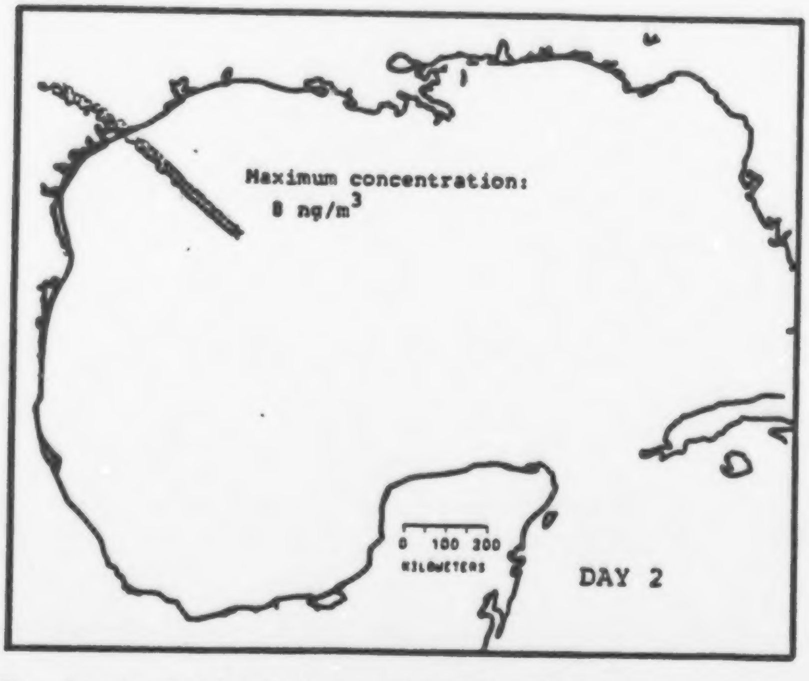
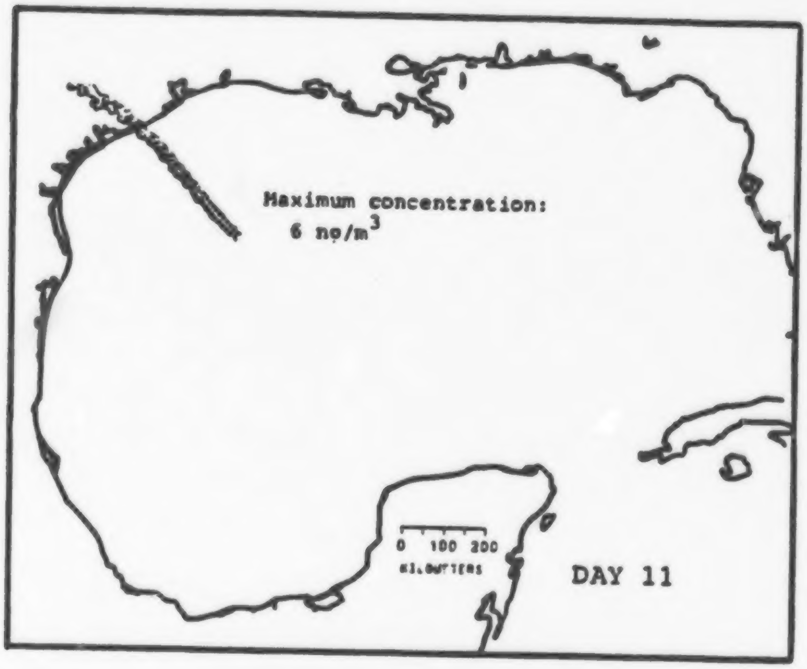


Exhibit G-8b Incineration plume concentration contours during and after an 11 day burn beginning April 1. Background contaminant levels are neglected in computing concentration values.



Contour intervals: 0.1,1,5,10,15 ng/m^3

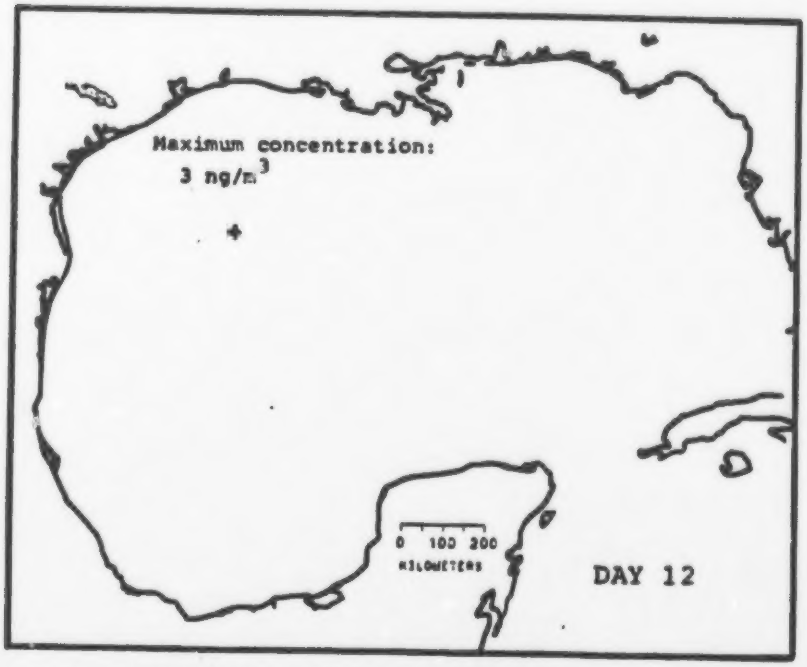
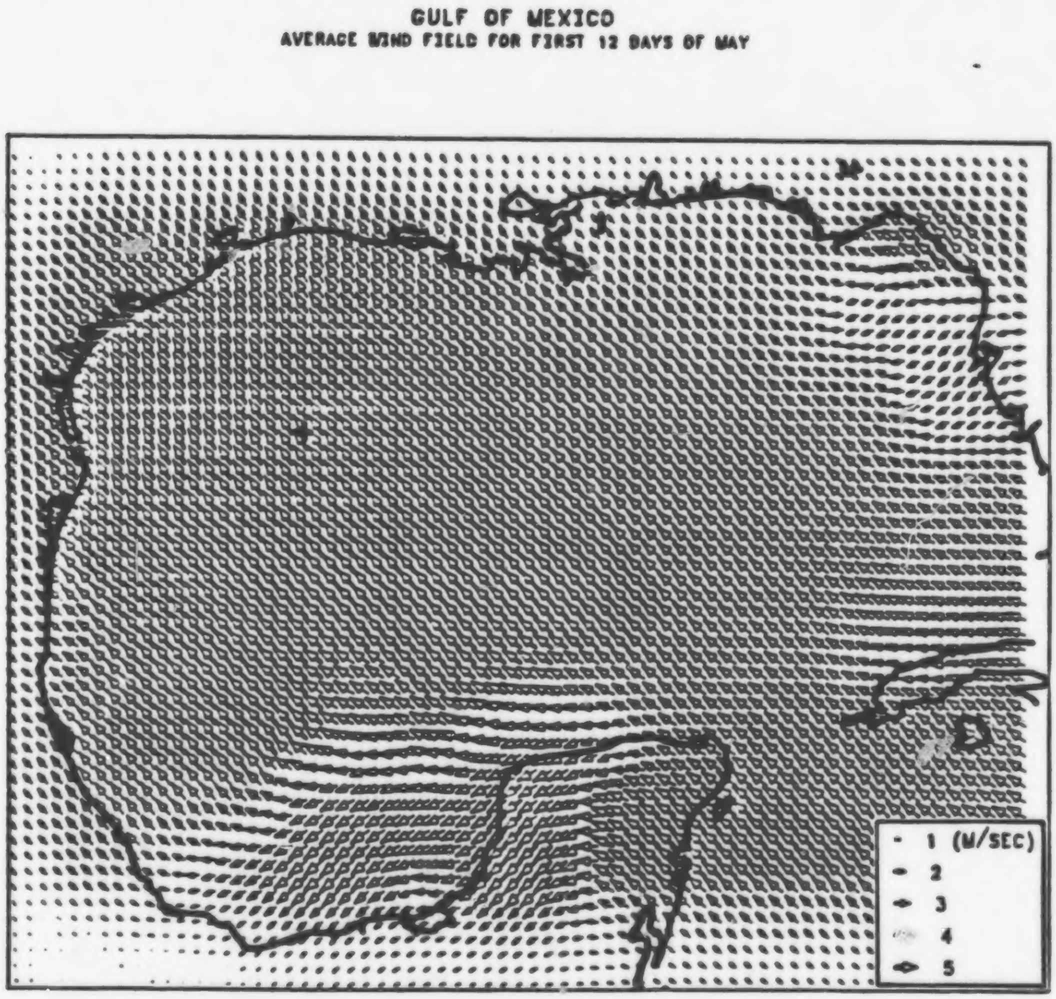


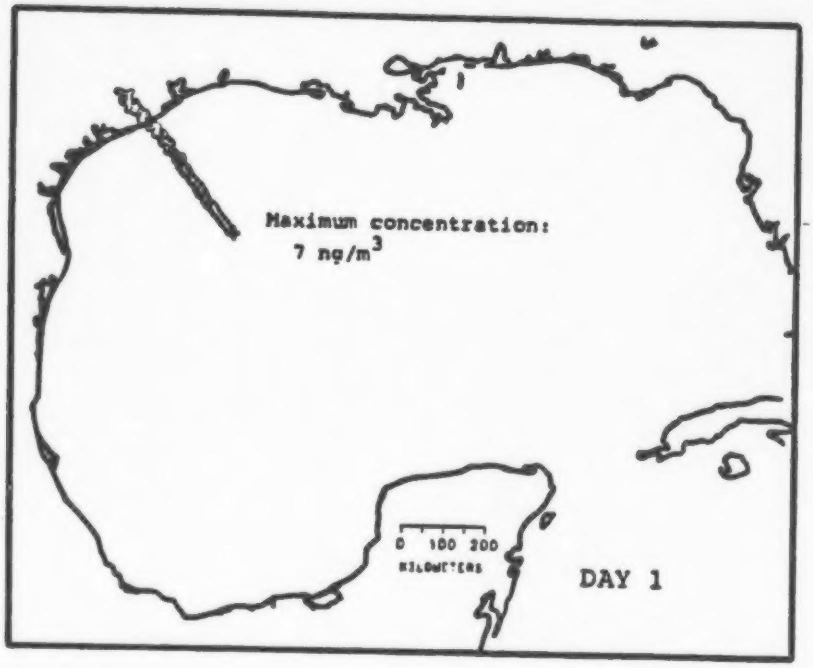
Exhibit G-8b cont. Incineration plume concentration contours during and after an 11 day burn beginning April 1. Background contaminant levels are neglected in computing concentration values.



GULF OF MEXICO

AVERAGE WIND FIELD FOR FIRST 12 DAYS OF MAY

Exhibit G-9a Average wind field for the first 12 days of May.



Contour intervals: 0.1,1,5,10,15 ng/m^3

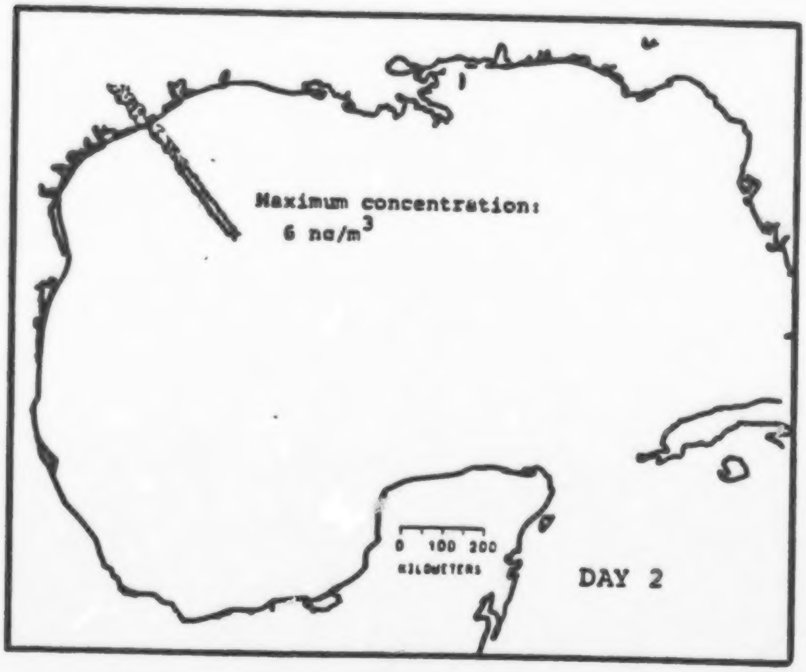
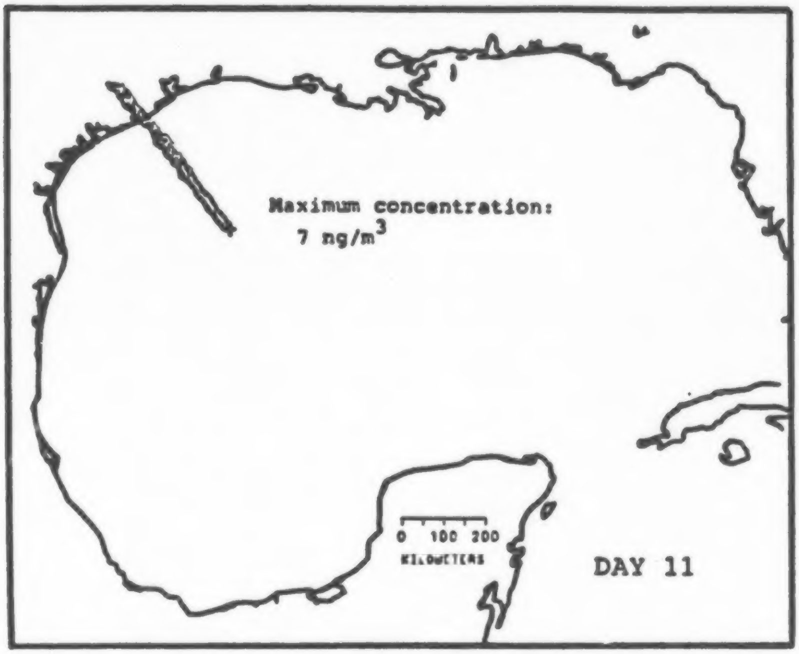


Exhibit G-9b Incineration plume concentration contours during and after an 11 day burn beginning May 1. Background contaminant levels are neglected in computing concentration values.



Contour intervals: 0.1,1,5,10,15 ng/m^3

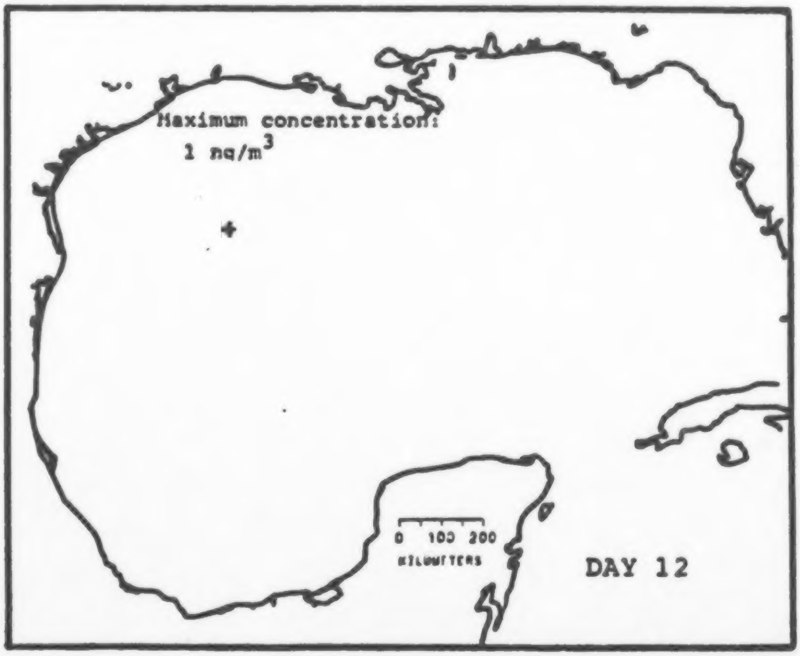


Exhibit G-9b cont. Incineration plume concentration contours during and after an 11 day burn beginning May 1. Background contaminant levels are neglected in computing concentration values.

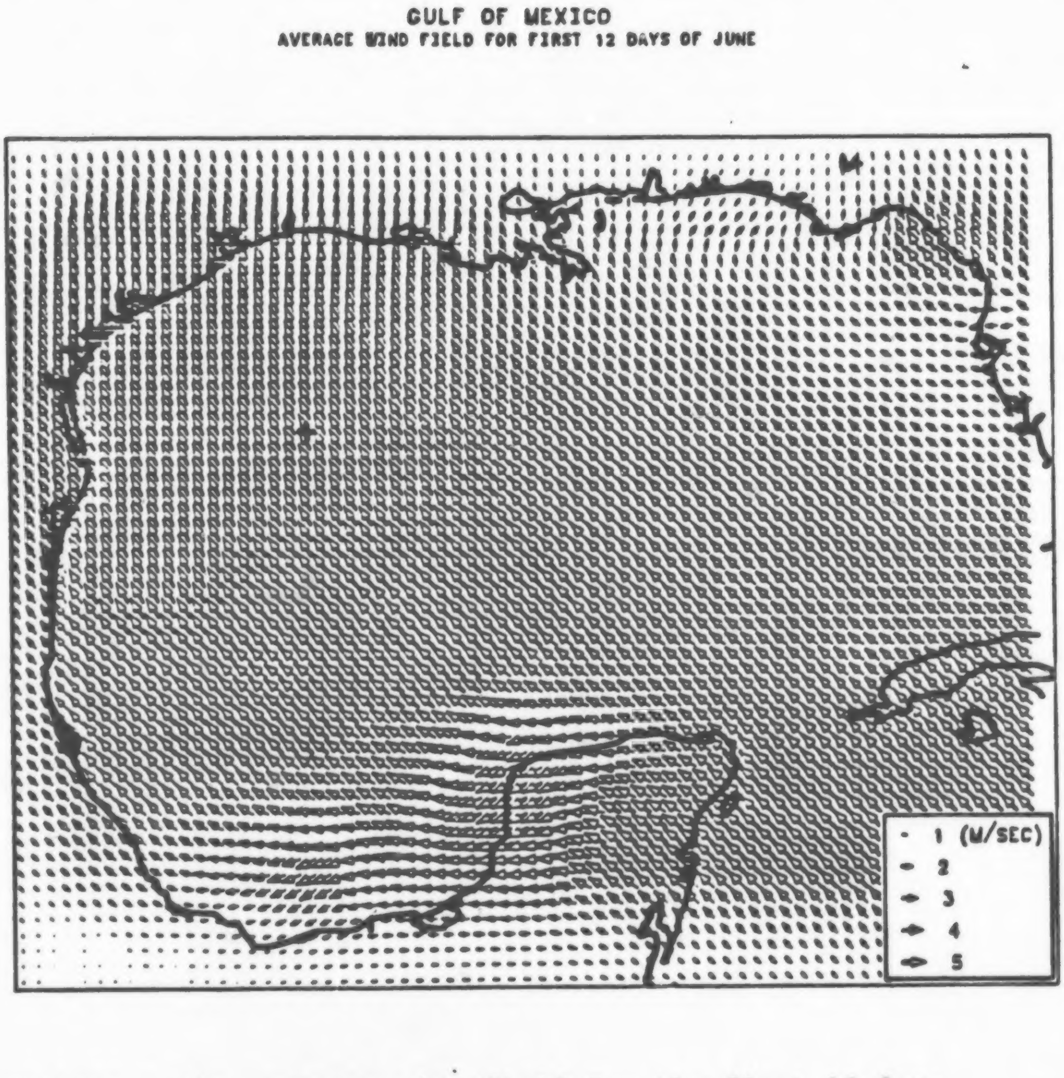
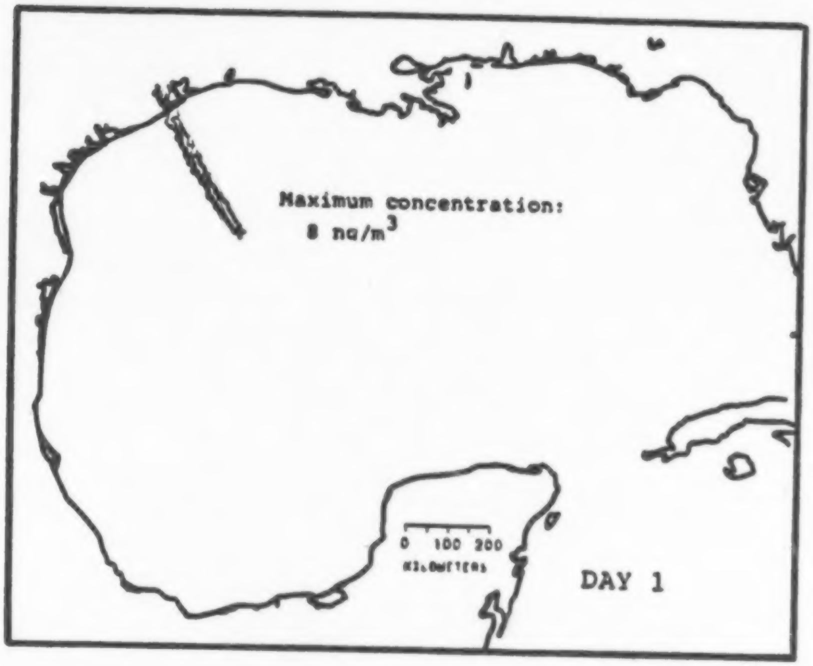


Exhibit G-10a Average wind field for the first 12 days of June.



Contour intervals: 0.1,1,5,10,15 ng/m^3

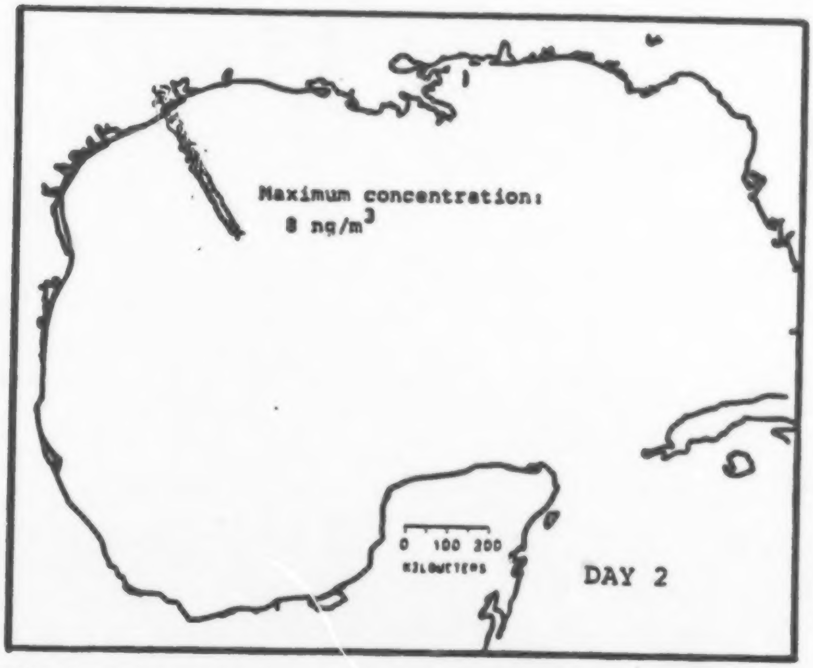
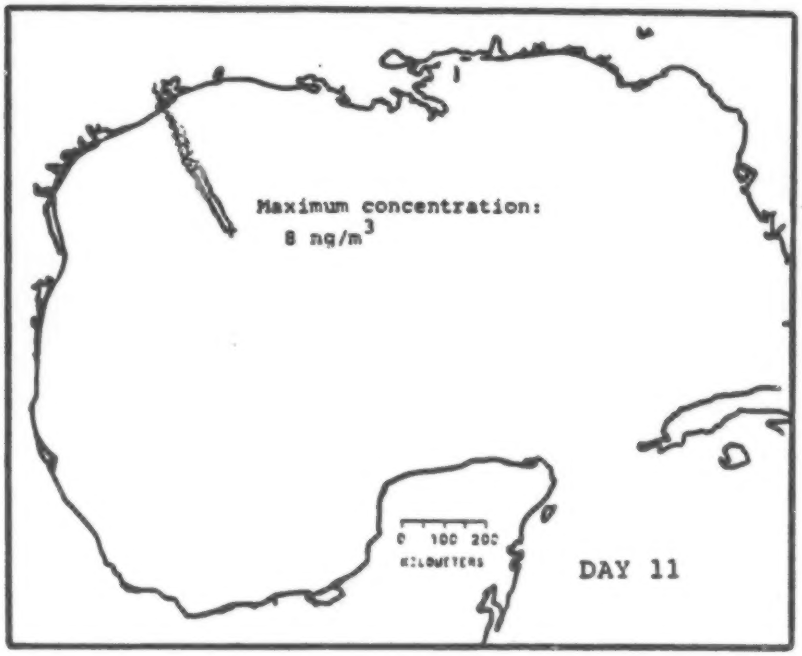


Exhibit G-10b Incineration plume concentration contours during and after an 11 day burn beginning June 1. Background contaminant levels are neglected in computing concentration values.



Contour intervals: 0.1,1,5,10,15 ng/m^3

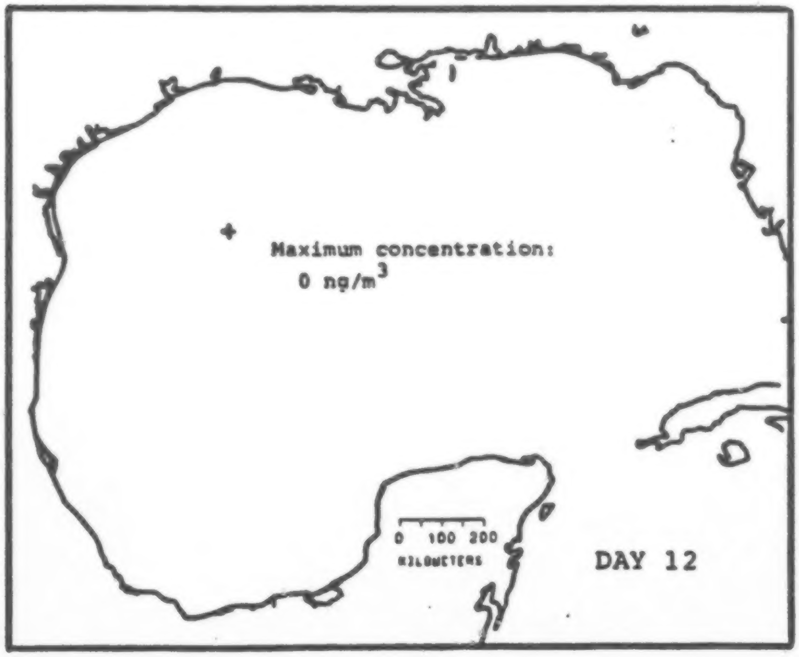


Exhibit G-10b cont. Incineration plume concentration contours during and after an 11 day burn beginning June 1. Background contaminant levels are neglected in computing concentration values.

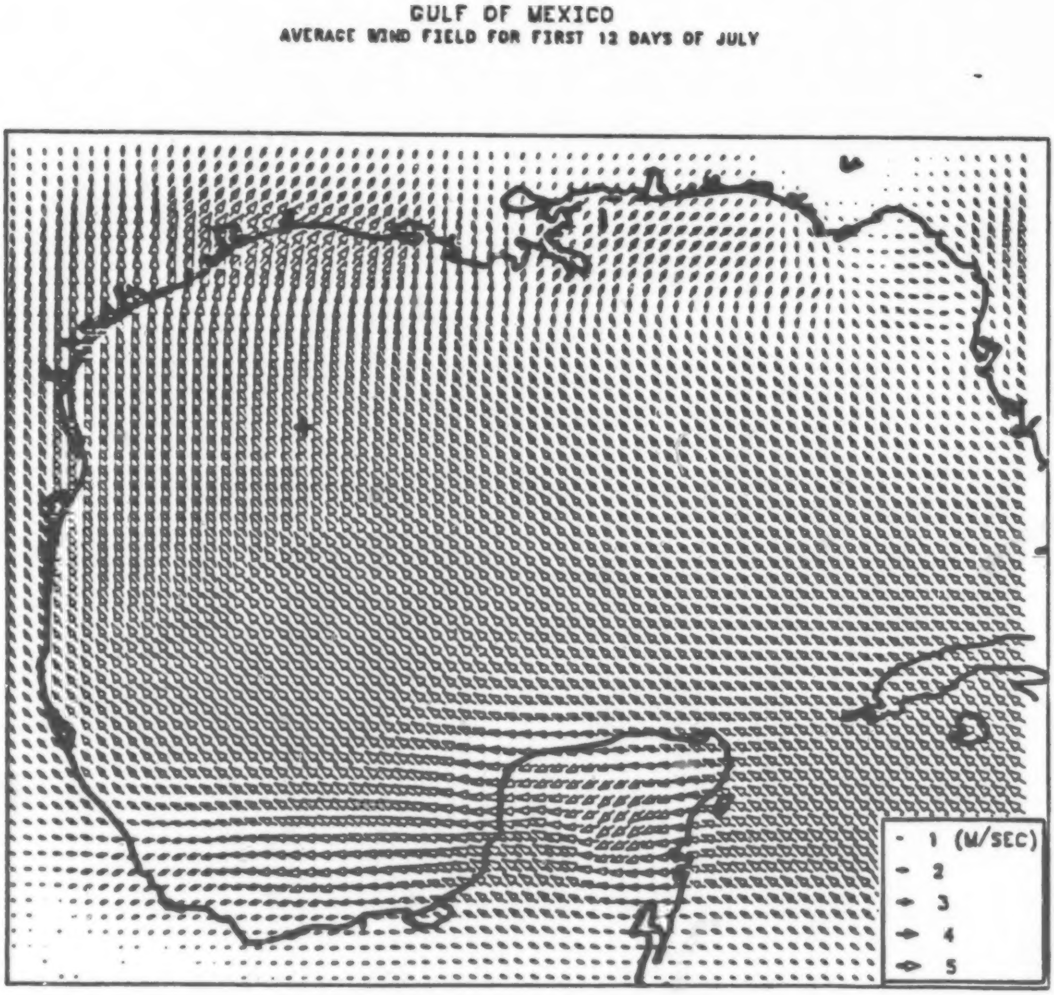
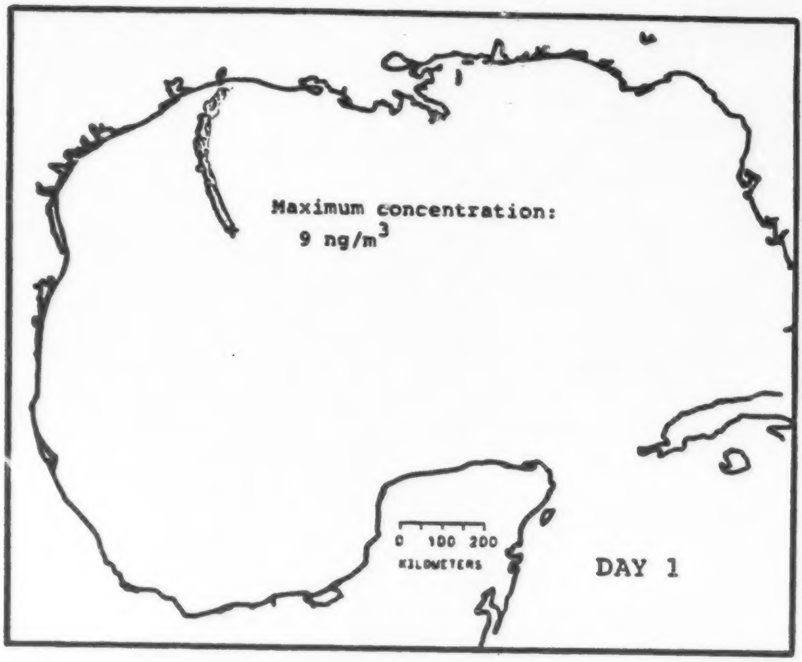


Exhibit G-11a Average wind field for the first 12 days of July.



Contour intervals: 0.1,1,5,10,15 ng/m^3

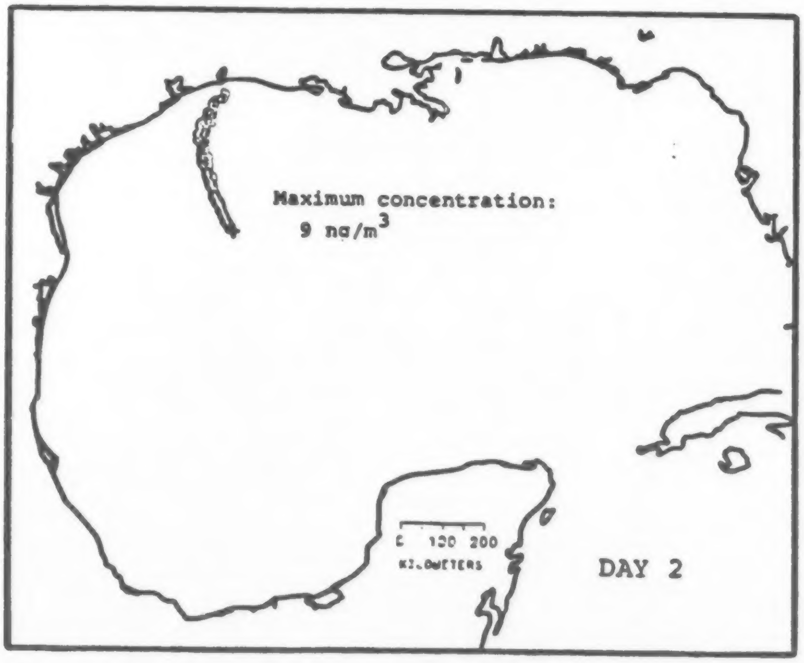
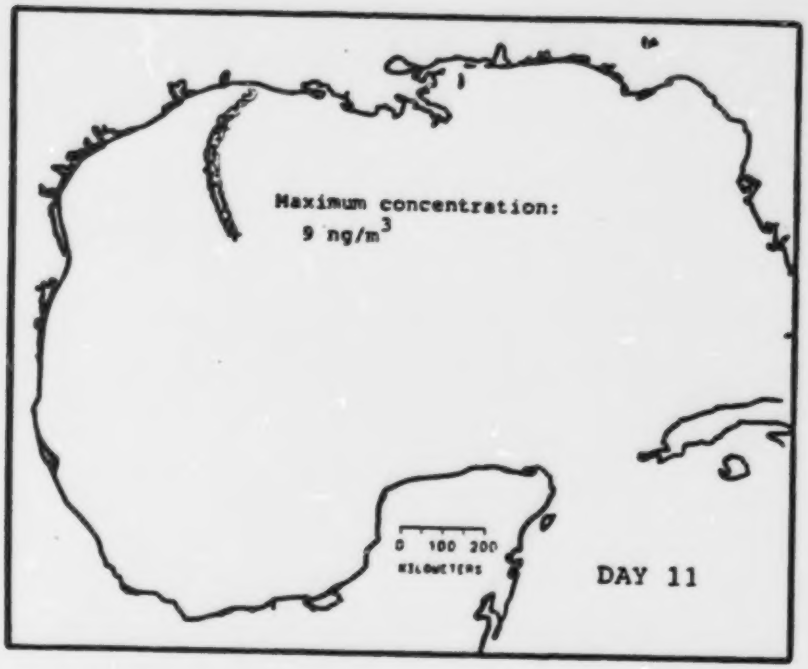


Exhibit G-11b Incineration plume concentration contours during and after an 11 day burn beginning July 1. Background contaminant levels are neglected in computing concentration values.



Contour intervals: 0.1,1,5,10,15 ng/m^3

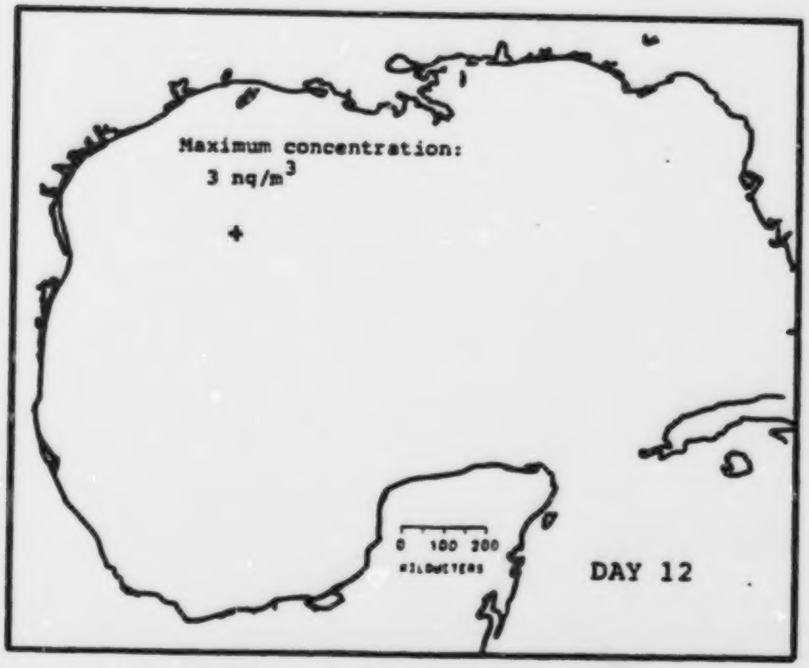


Exhibit G-11b cont. Incineration plume concentration contours during and after an 11 day burn beginning July 1. Background contaminant levels are neglected in computing concentration values.

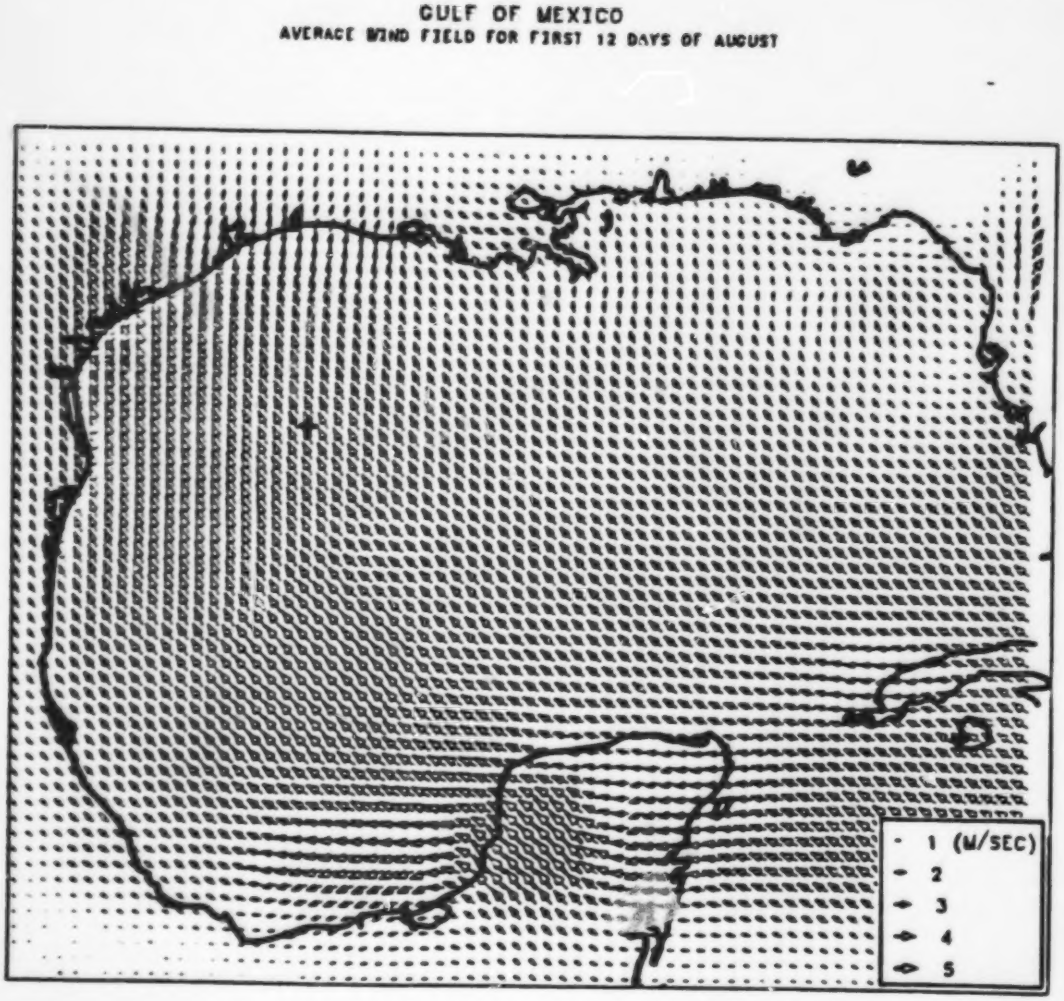
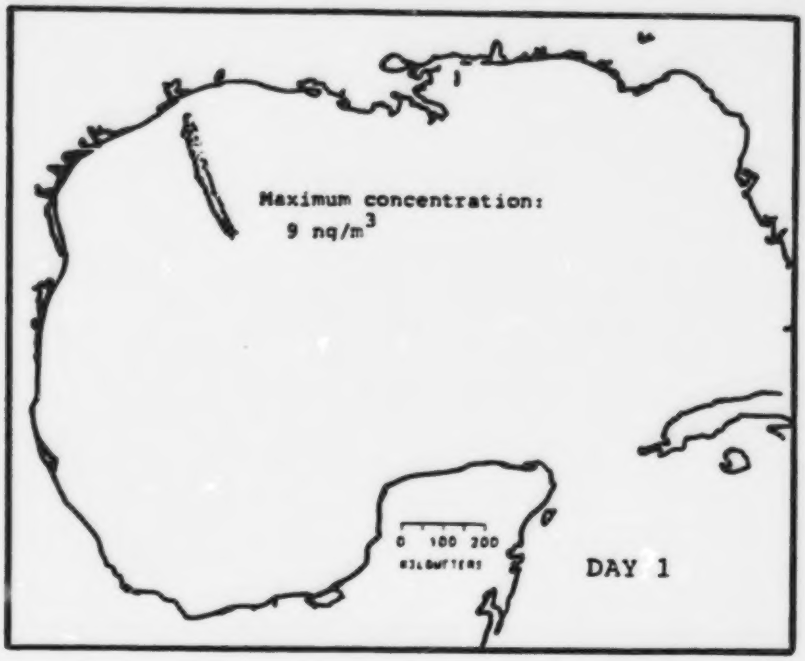


Exhibit G-12a Average wind field for the first 12 days of August.



Contour intervals: 0.1,1,5,10,15 ng/m^3

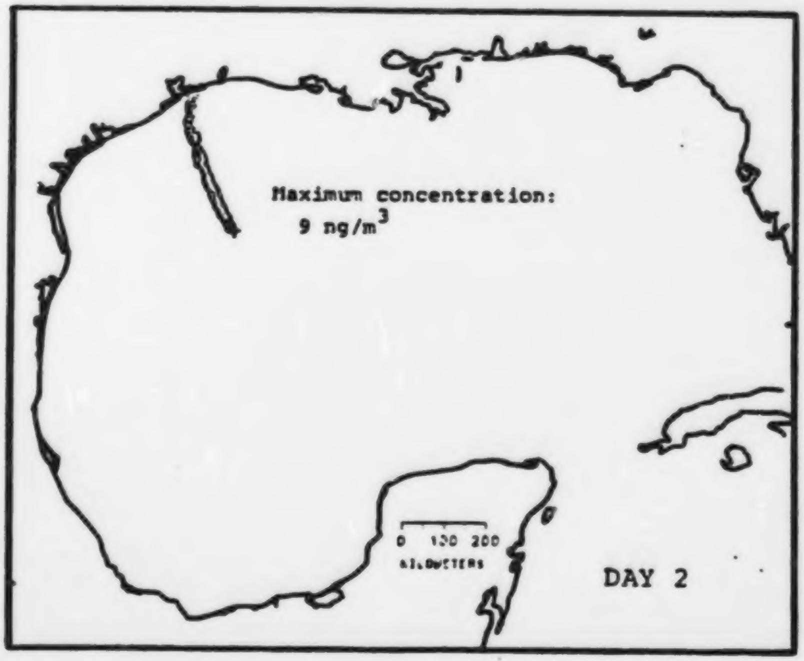
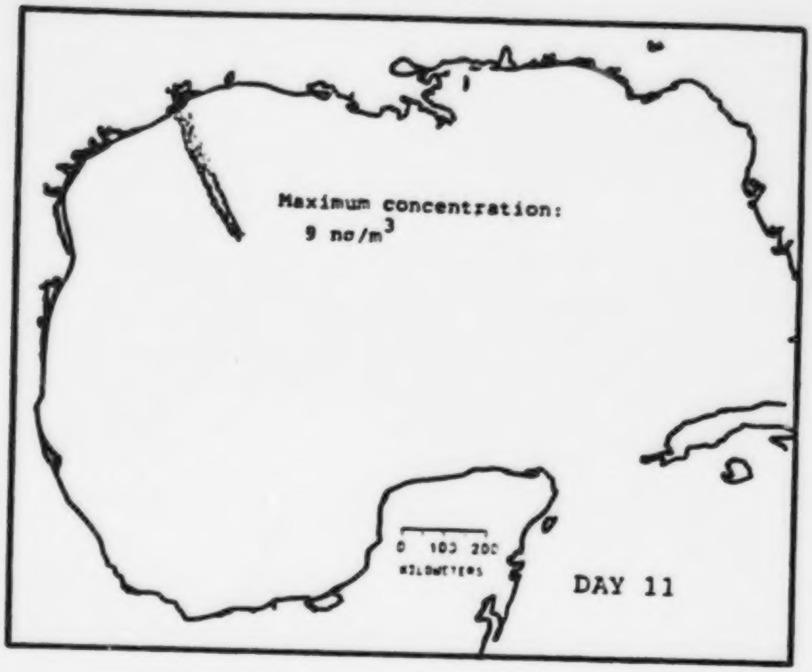


Exhibit G-12b Incineration plume concentration contours during and after an 11 day burn beginning August 1. Background contaminant levels are neglected in computing concentration values.



Contour intervals: 0.1,1,5,10,15 ng/m^3

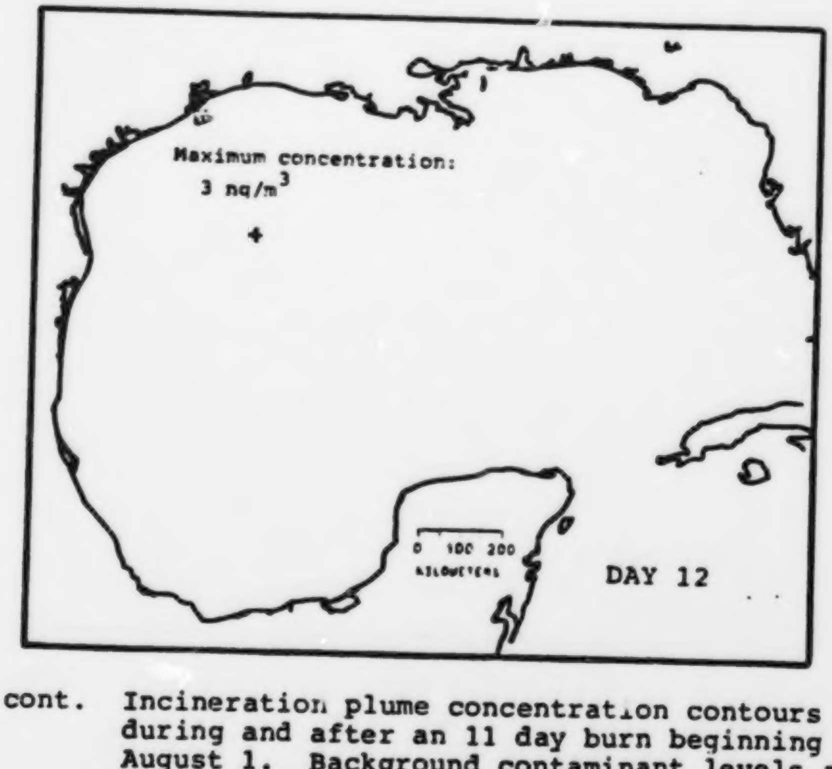


Exhibit G-12b cont. Incineration plume concentration contours during and after an 11 day burn beginning August 1. Background contaminant levels are neglected in computing concentration values.

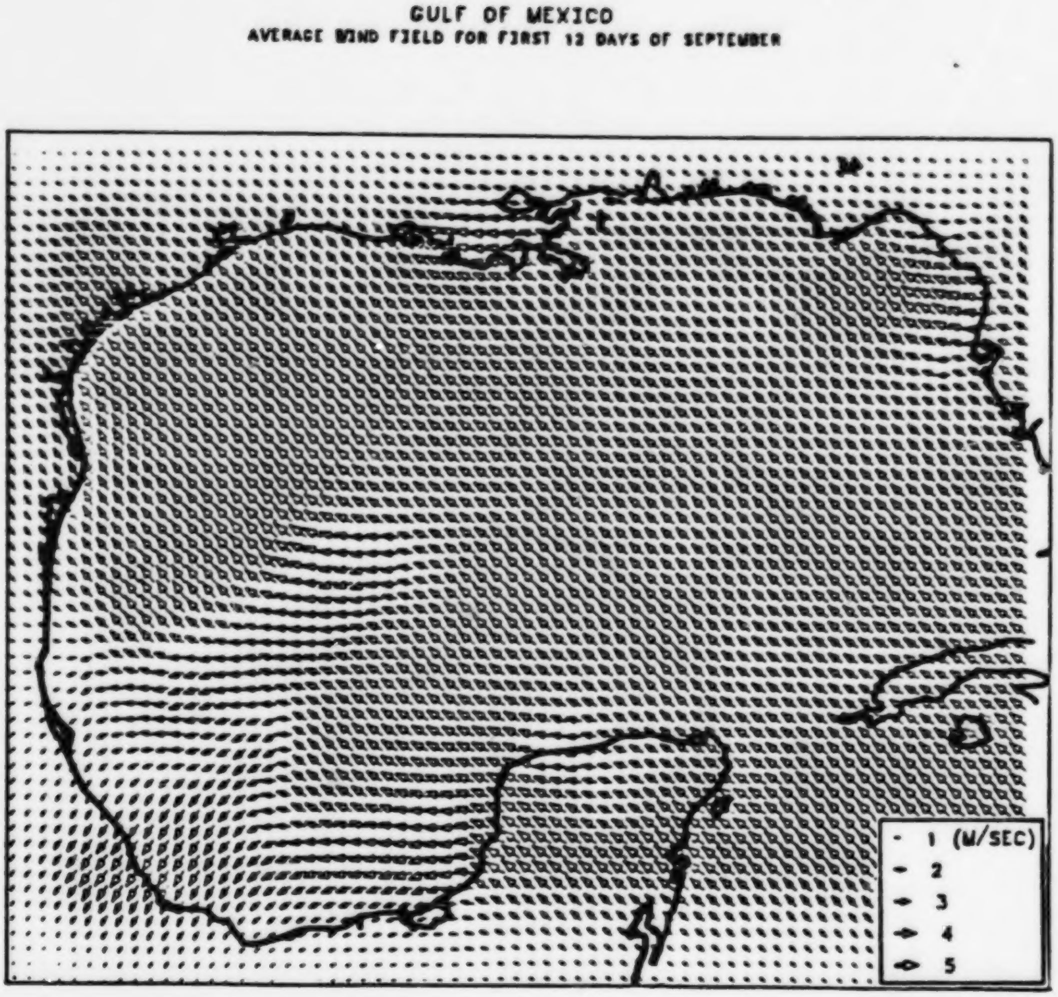
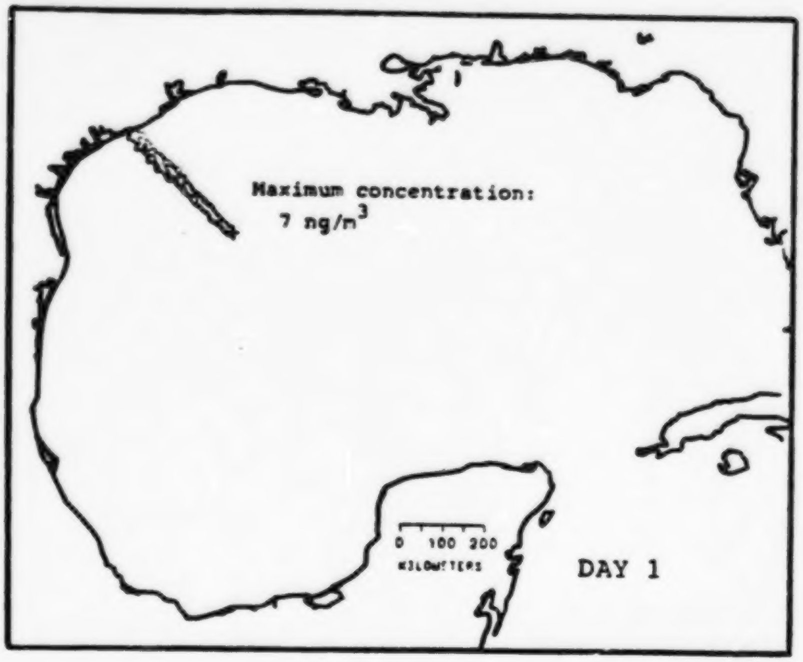


Exhibit G-13a Average wind field for the first 12 days of September.



Contour intervals: 0.1,1,5,10,15 ng/m^3

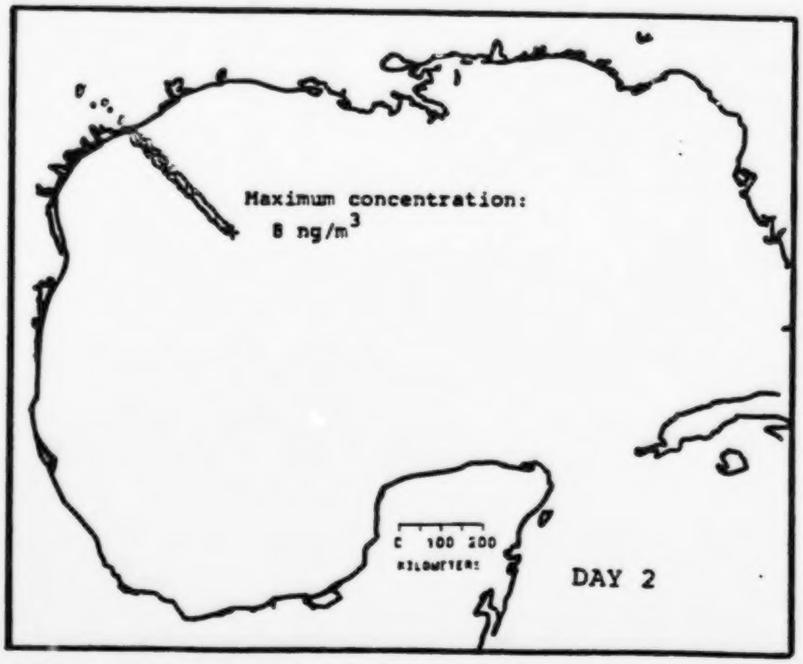
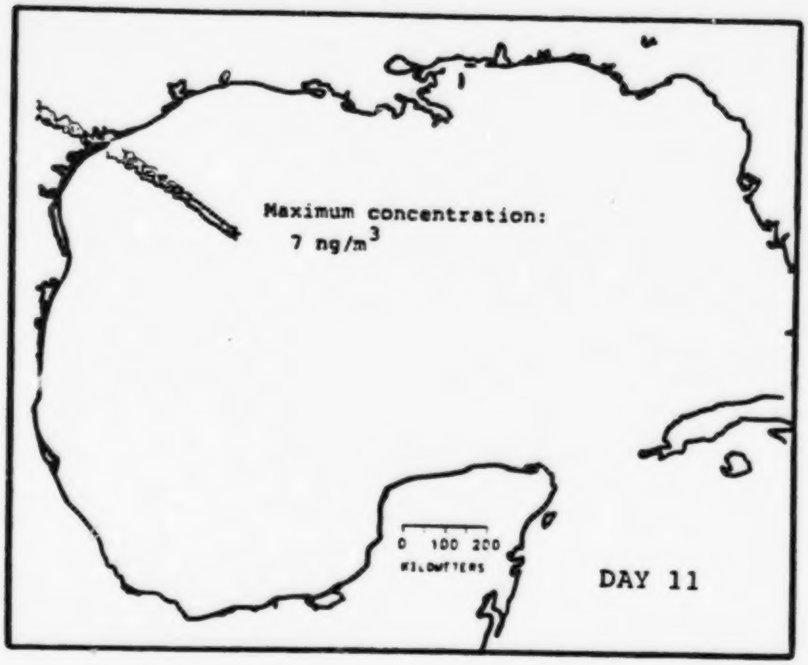


Exhibit G-13b Incineration plume concentration contours during and after an 11 day burn beginning September 1. Background contaminant levels are neglected in computing concentration values.



Contour intervals: 0.1,1,5,10,15 ng/m^3

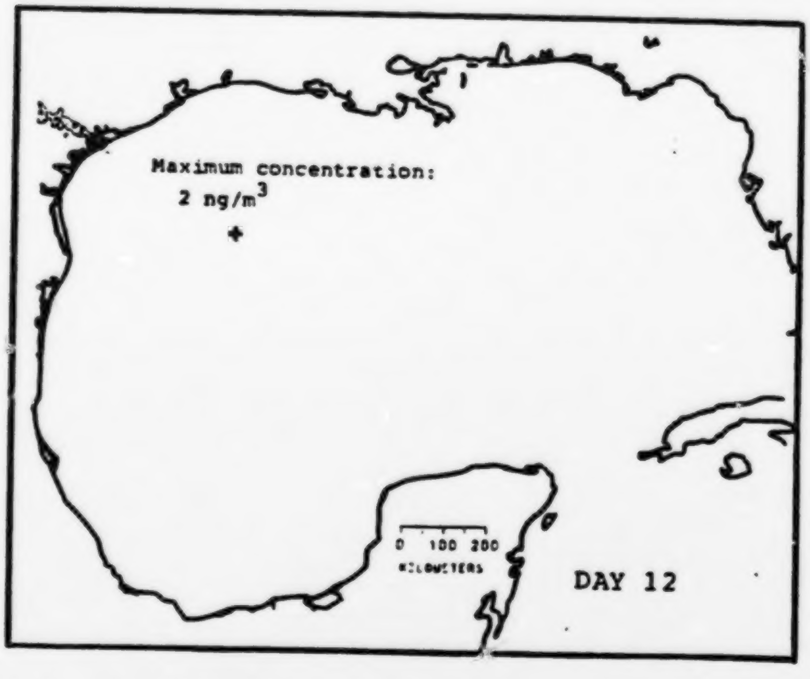


Exhibit G-13b cont. Incineration plume concentration contours during and after an 11 day burn beginning September 1. Background contaminant levels are neglected in computing concentration values.

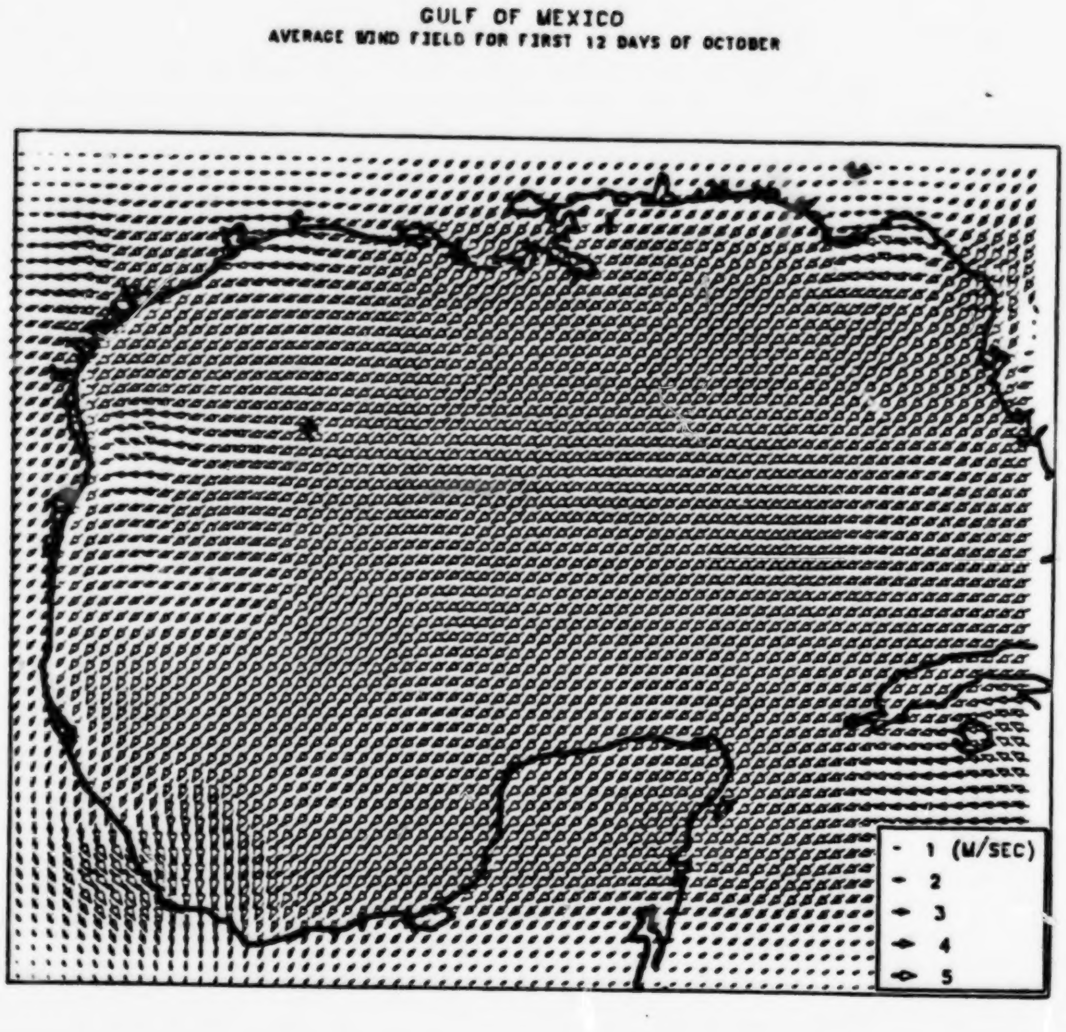
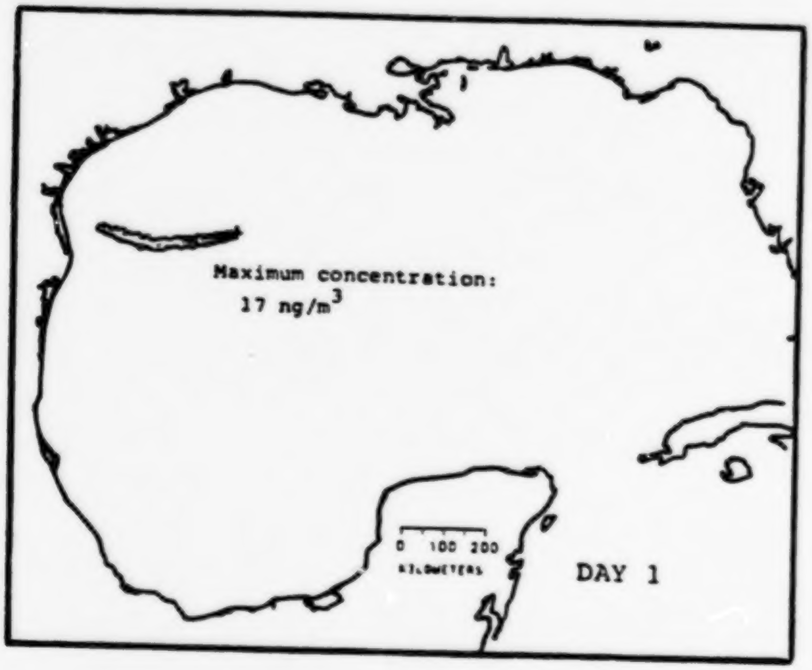


Exhibit G-14a Average wind field for the first 12 days of October.



Contour intervals: 0.1,1,5,10,15 ng/m^3

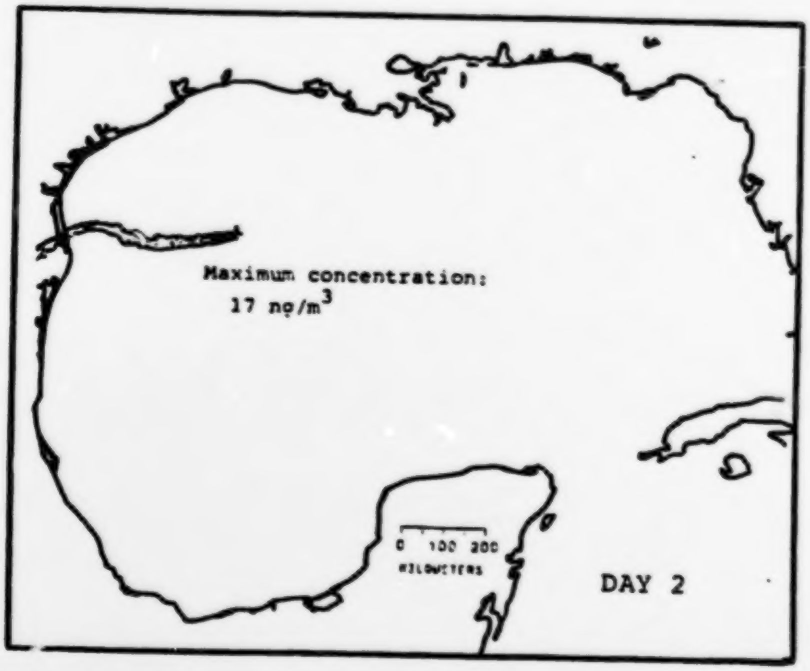
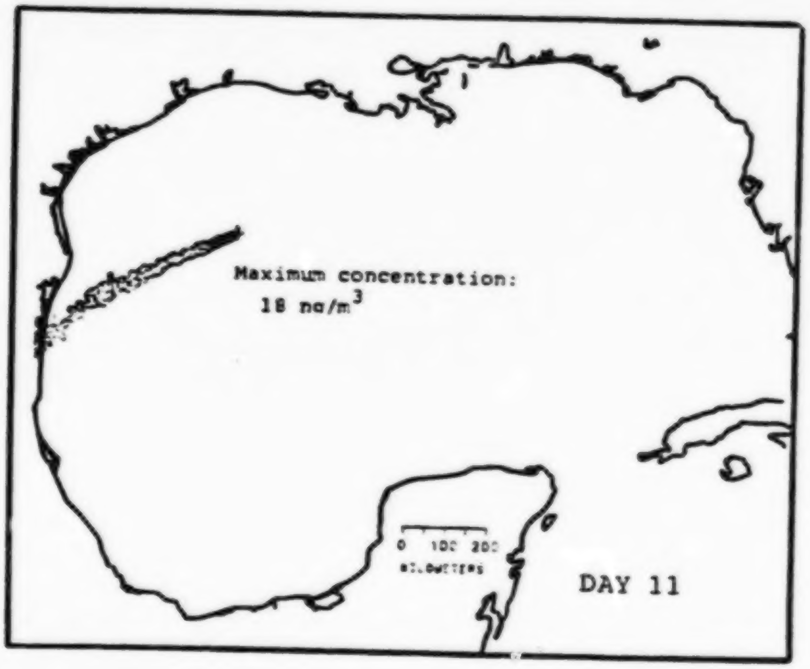


Exhibit G-14b Incineration plume concentration contours during and after an 11 day burn beginning October 1. Background contaminant levels are neglected in computing concentration values.



Contour intervals: 0.1,1,5,10,15 ng/m^3

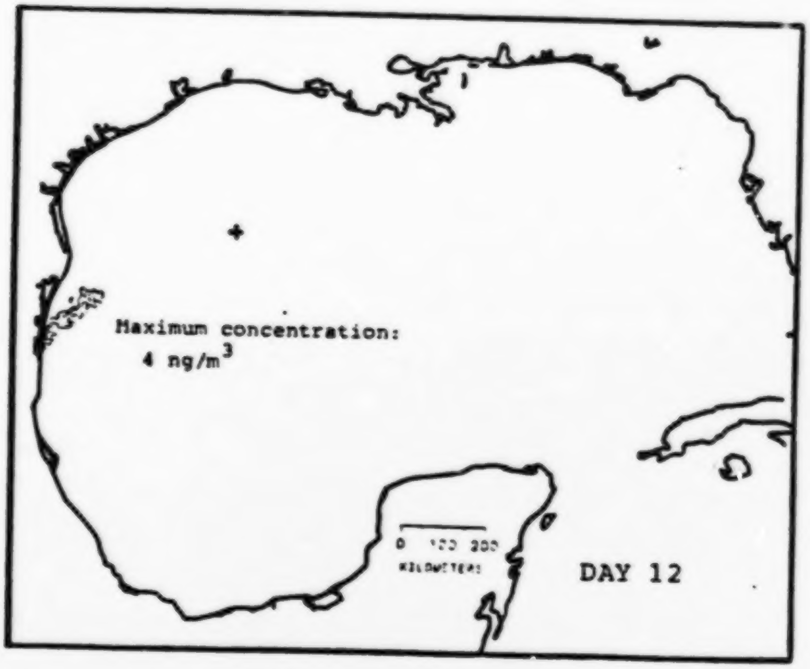


Exhibit G-14b cont. Incineration plume concentration contours during and after an 11 day burn beginning October 1. Background contaminant levels are neglected in computing concentration values.

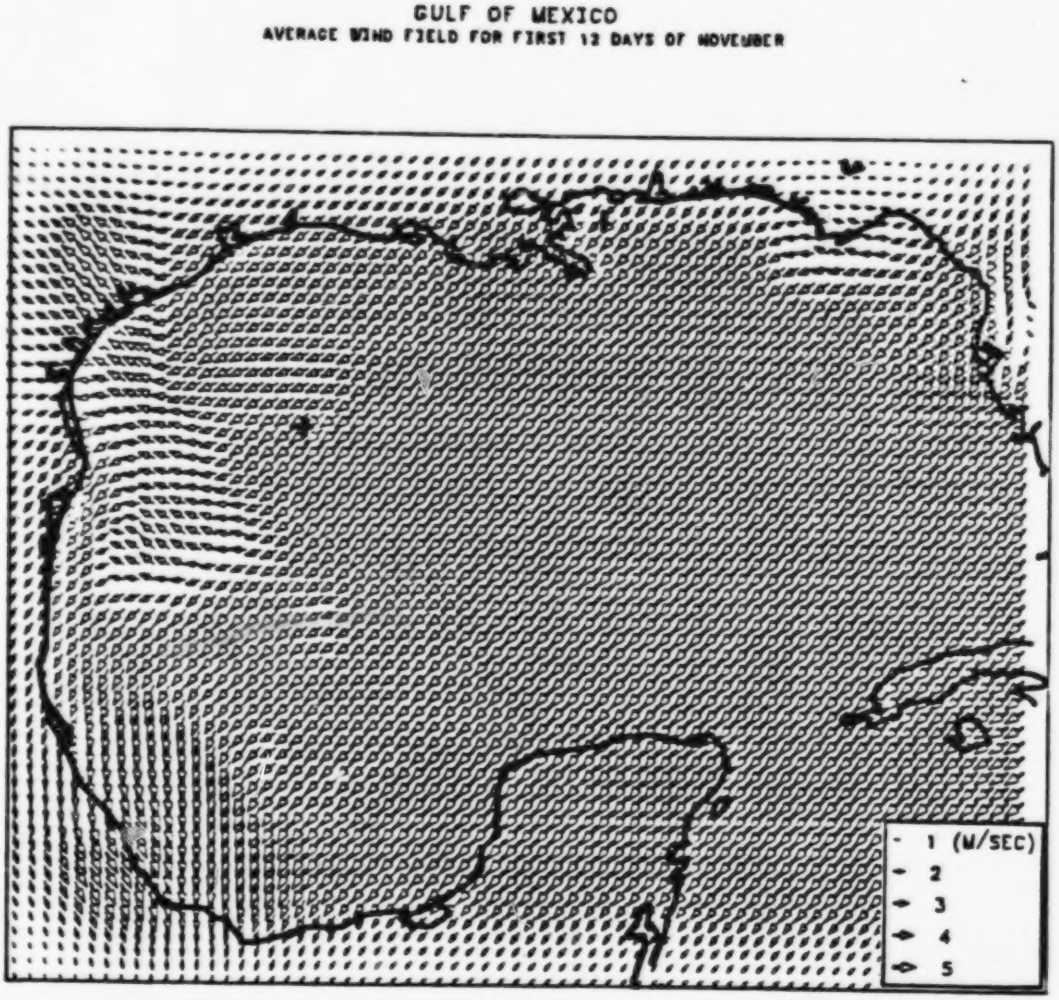
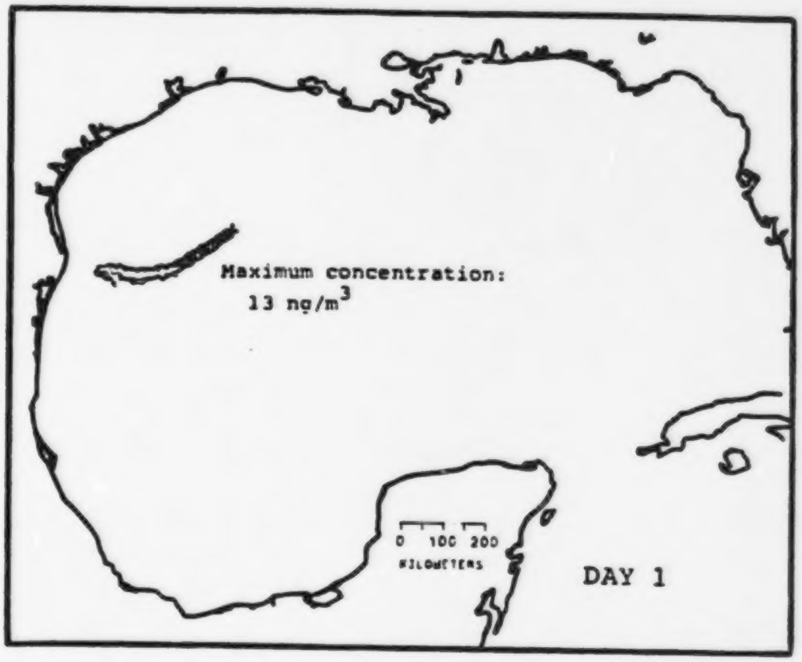


Exhibit G-15a Average wind field for the first 12 days of November.



Contour intervals: 0.1,1,5,10,15 ng/m^3

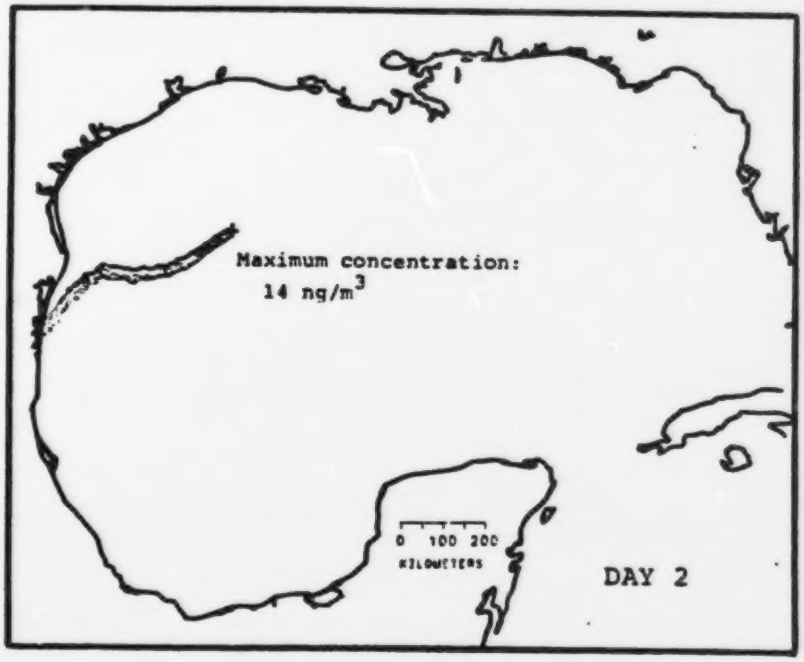
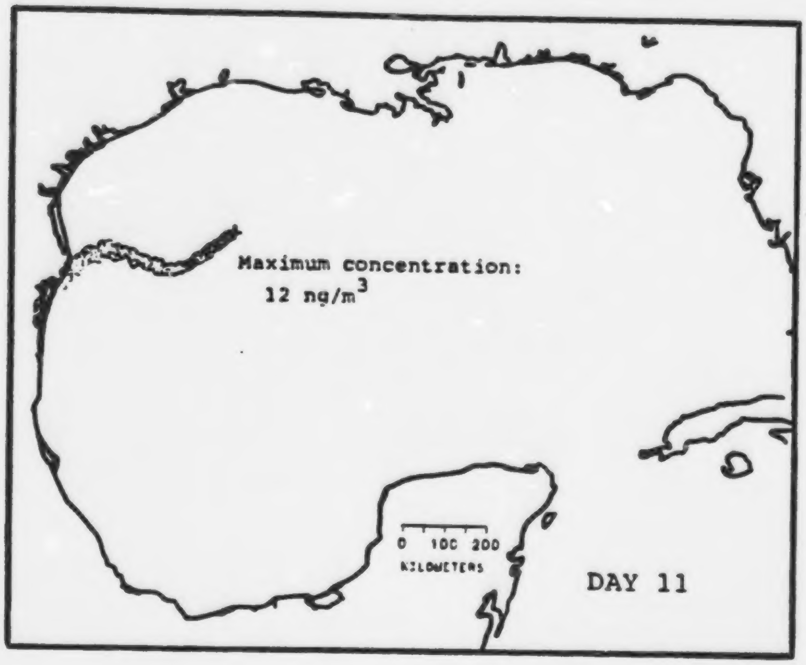


Exhibit G-15b Incineration plume concentration contours during and after an 11 day burn beginning November 1. Background contaminant levels are neglected in computing concentration values.



Contour intervals: 0.1,1,5,10,15 ng/m^3

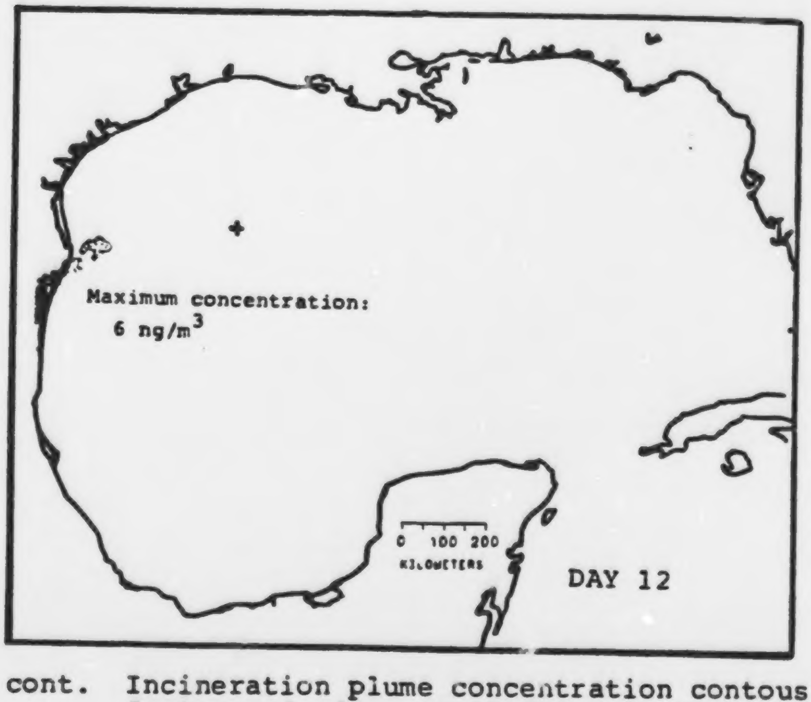


Exhibit G-15b cont. Incineration plume concentration contous during and after an 11 day burn beginning November 1. Background contaminant levels are neglected in computing concentration values.

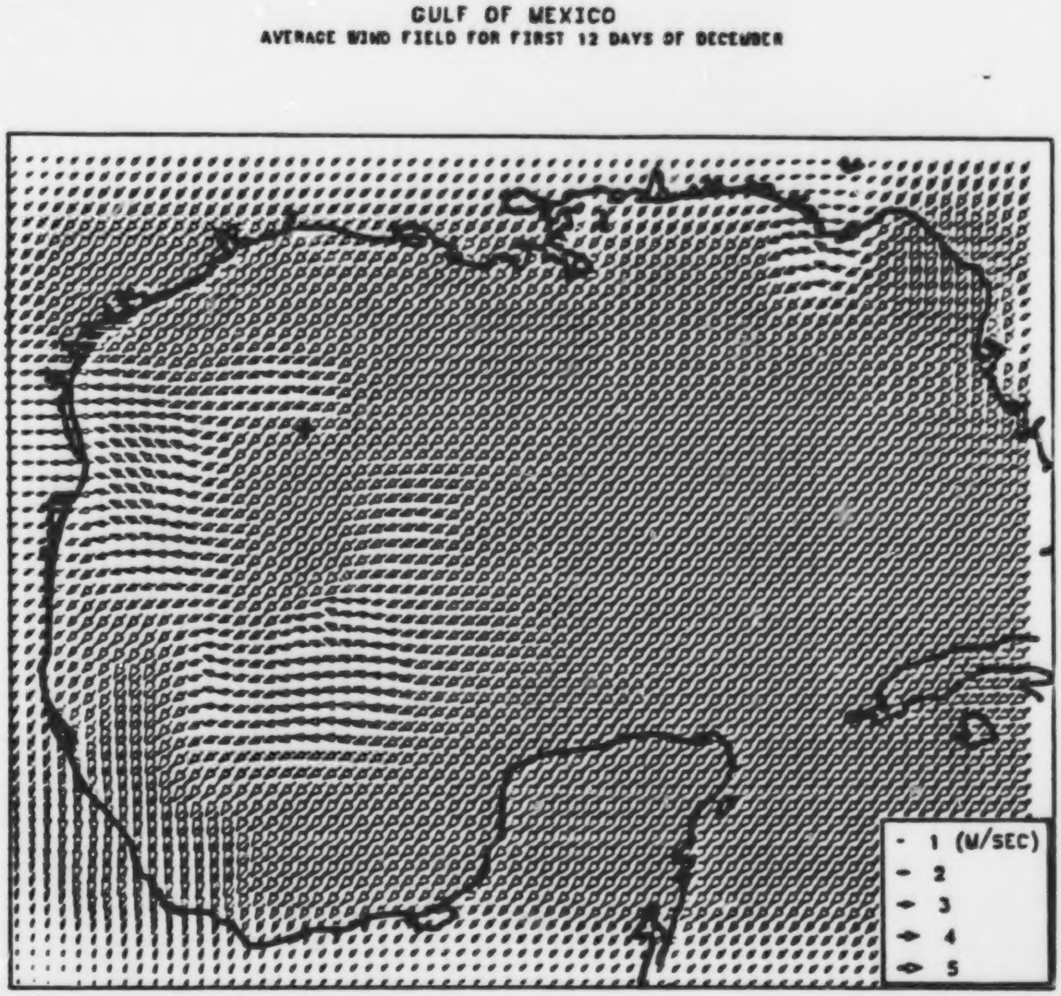
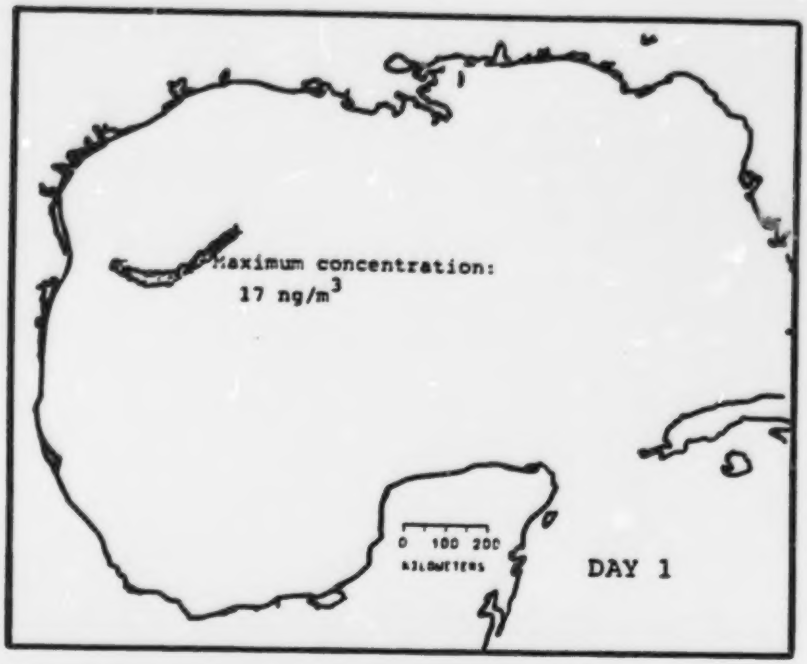


Exhibit G-16a Average wind field for the first 12 days of December.



Contour intervals: 01,1,5,10,15 ng/m3

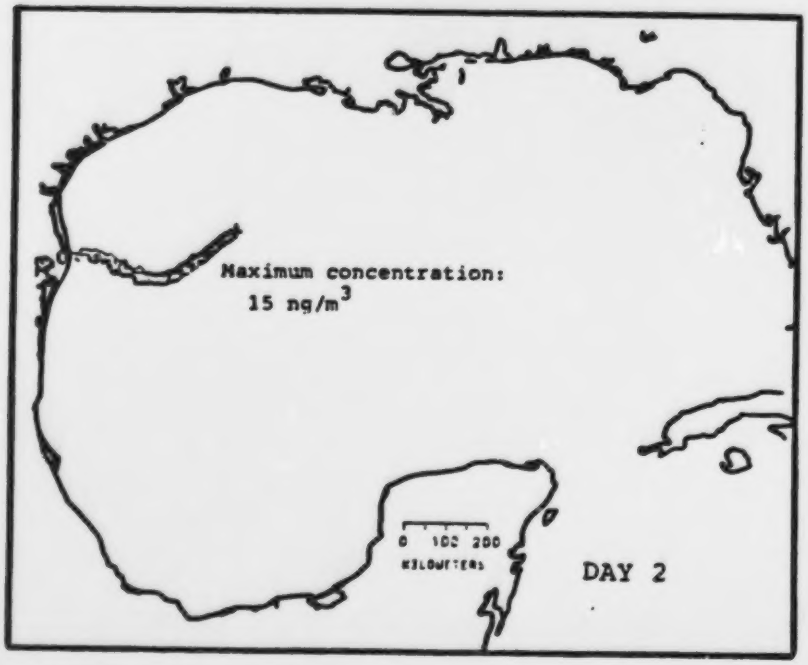
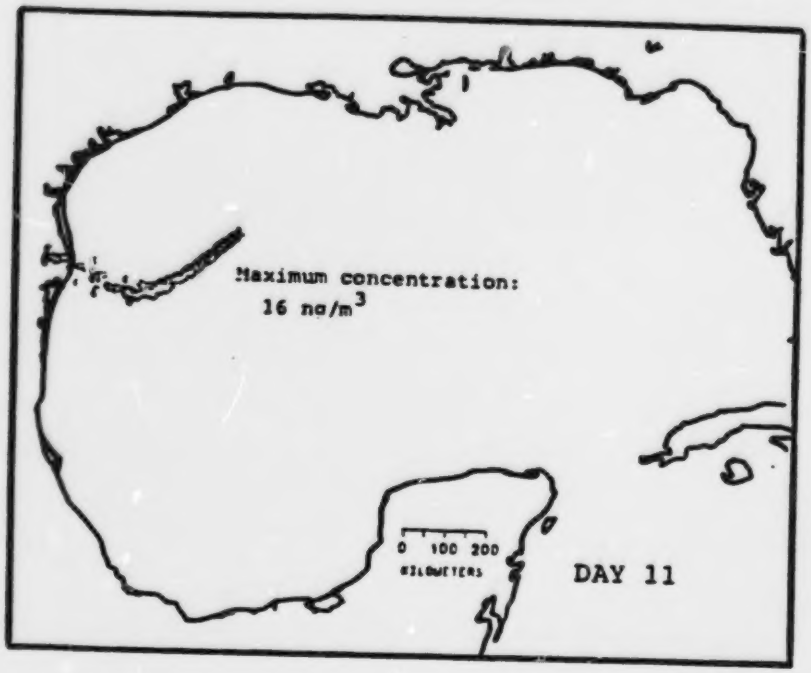


Exhibit G-16b Incineration plume concentration contours during and after an 11 day burn beginning December 1. Background contaminant levels are neglected in computing concentration levels.



Contour intervals: 0.1,1,5,10,15 ng/m^3

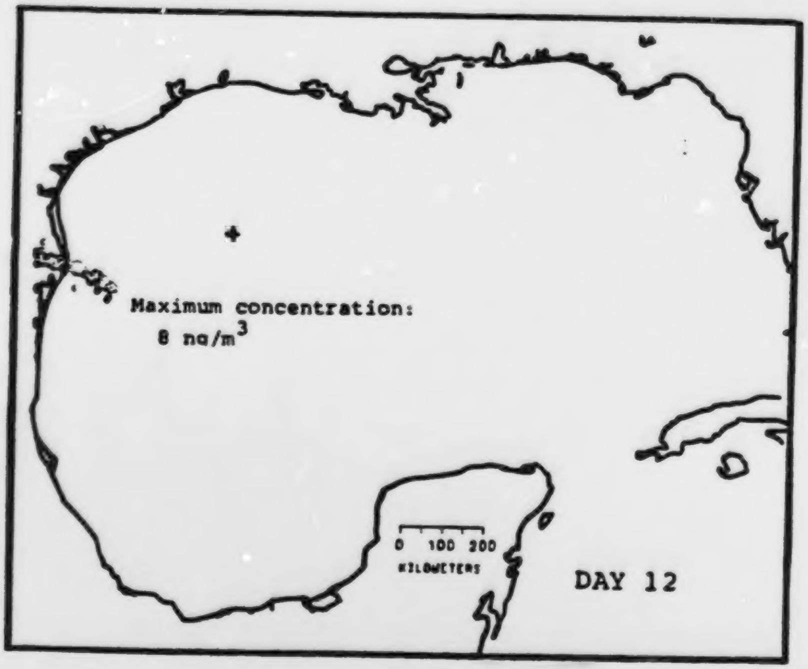
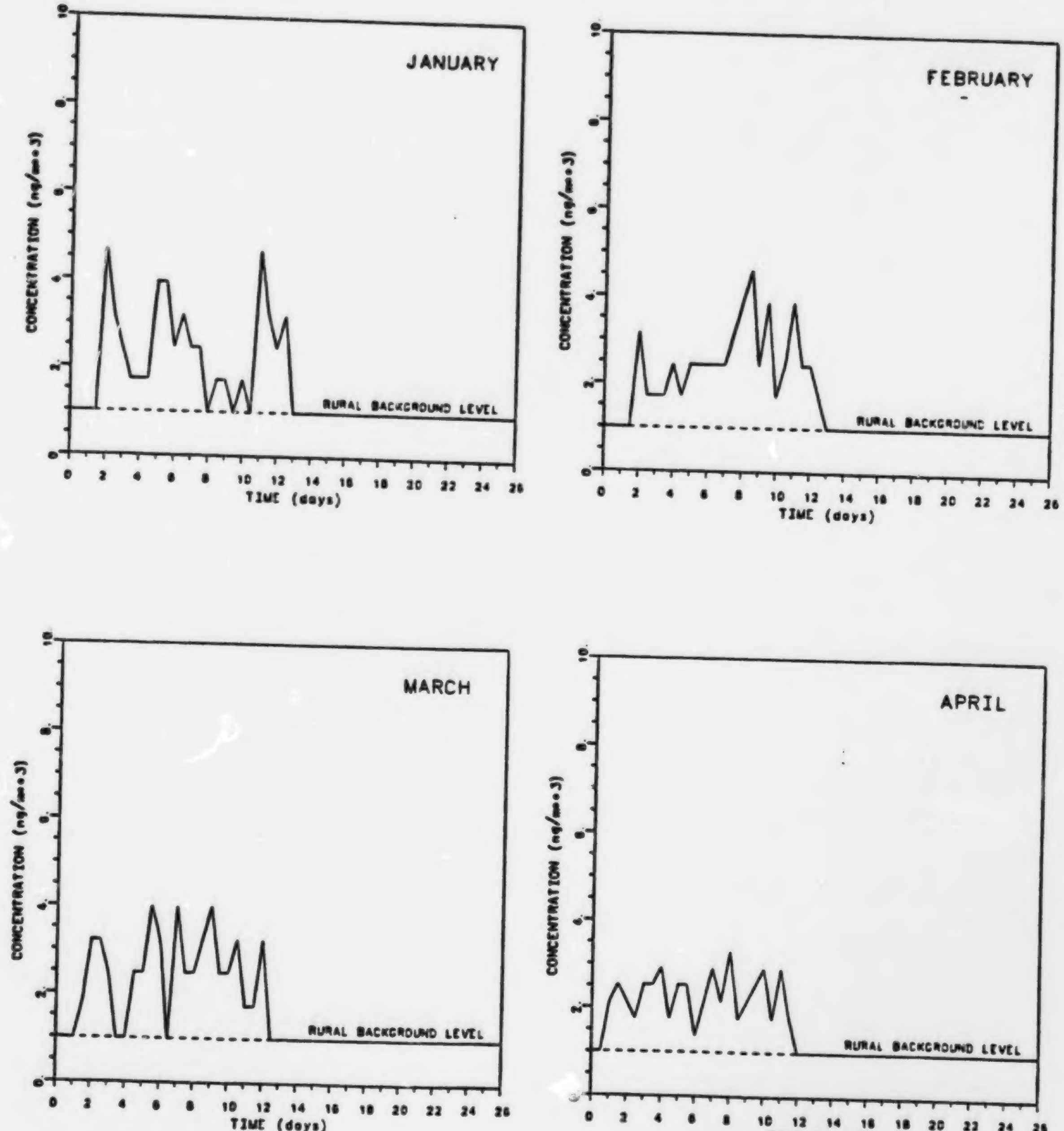
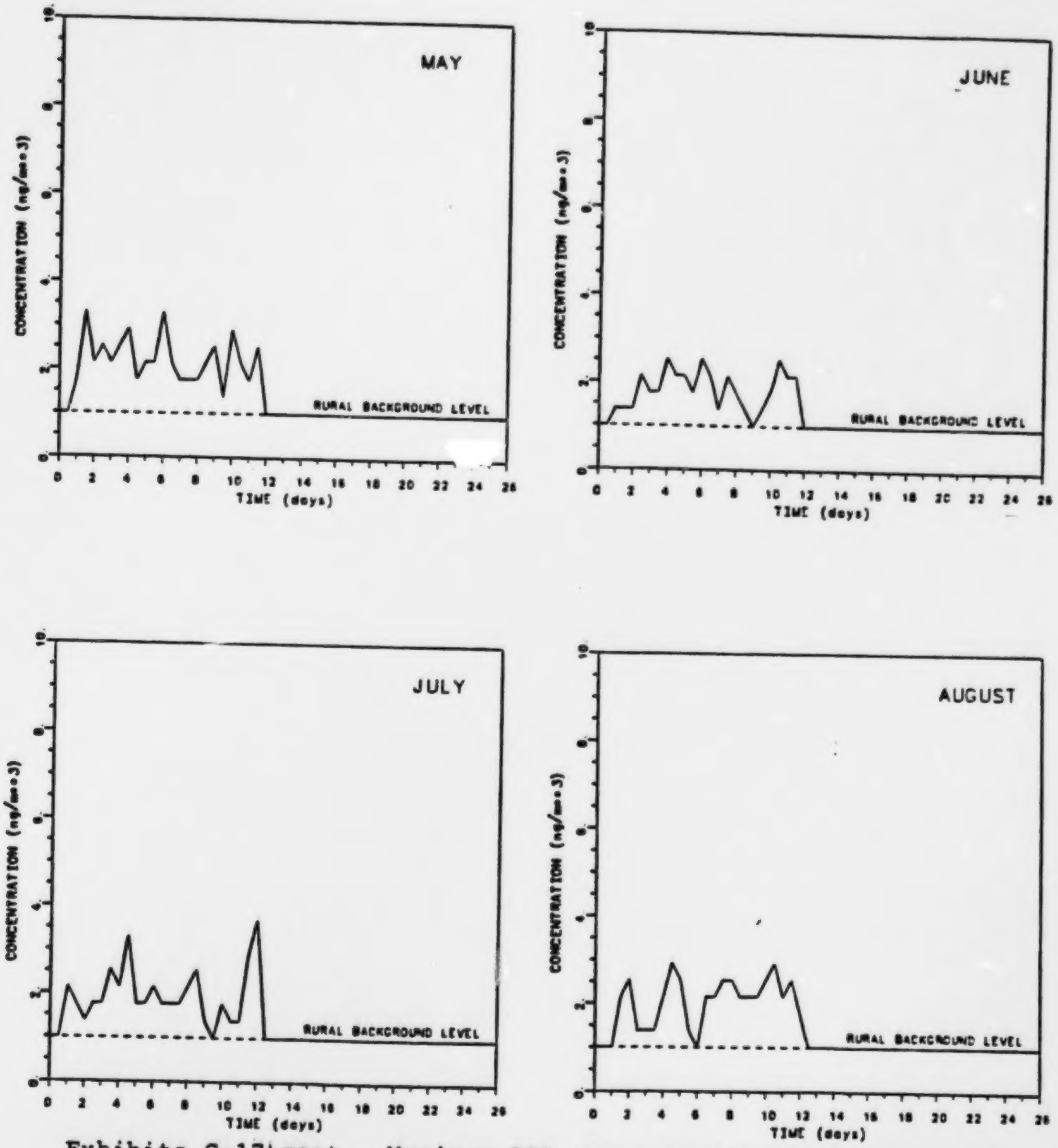


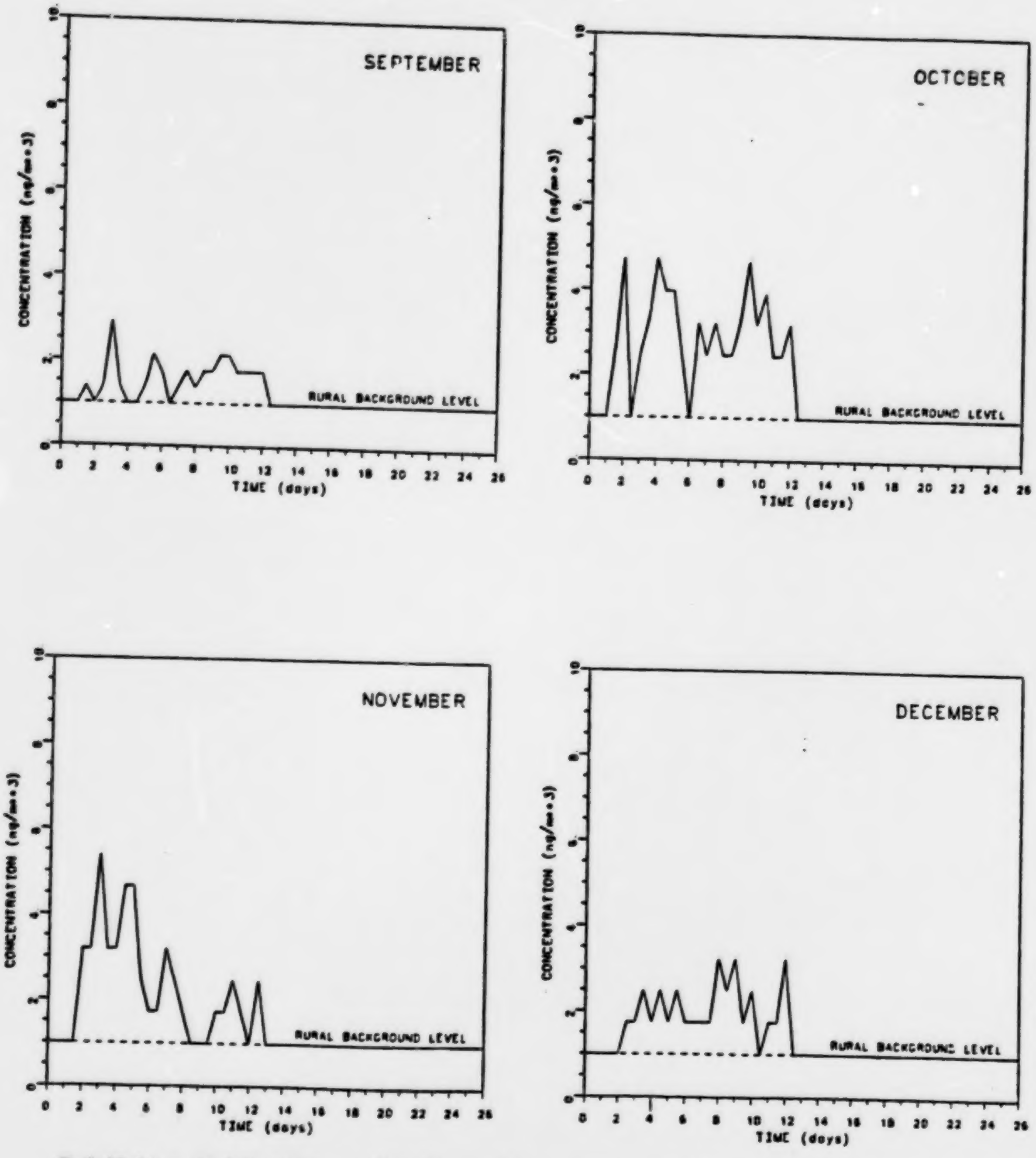
Exhibit G-16b cont. Incineration plume concentration contours during and after an 11 day burn beginning December 1. Background contaminant levels are neglected in computing concentration levels.



Exhibits G-17a Maximum PCB concentration time series at the coast due to 1 kg/hr incinerator stack source strength (a) January (b) February (c) March (d) April (e) May (f) June (g) July (h) August (i) September (j)October (k) November (l) December



Exhibits G-17b cont. Maximum PCB concentration time series at the coast due to 1 kg/hr incinerator stack source strength (a) January (b) February (c) March (d) April (e) May (f) June (g) July (h) August (i) September (j) October (k) November (l) December



Exhibits G-17 cont. Maximum PCB concentration time series at the coast due to 1 kg/hr incinerator stack source strength (a) January (b) February (c) March (d) April (e) May (f) June (g) July (h) August (i) September (j) October (k) November (l) December

Exhibit G-18. Number of rain days occurring during each PCB incineration campaign simulation.

Month	Number of Rain Days	Probability of Rain
January	4	0.23
February	3	0.23
March	1	0.13
April	0	0.13
May	0	0.13
June	1	0.23
July	4	0.23
August	3	0.23
September	4	0.27
October	2	0.27
November	4	0.27
December	6	0.23

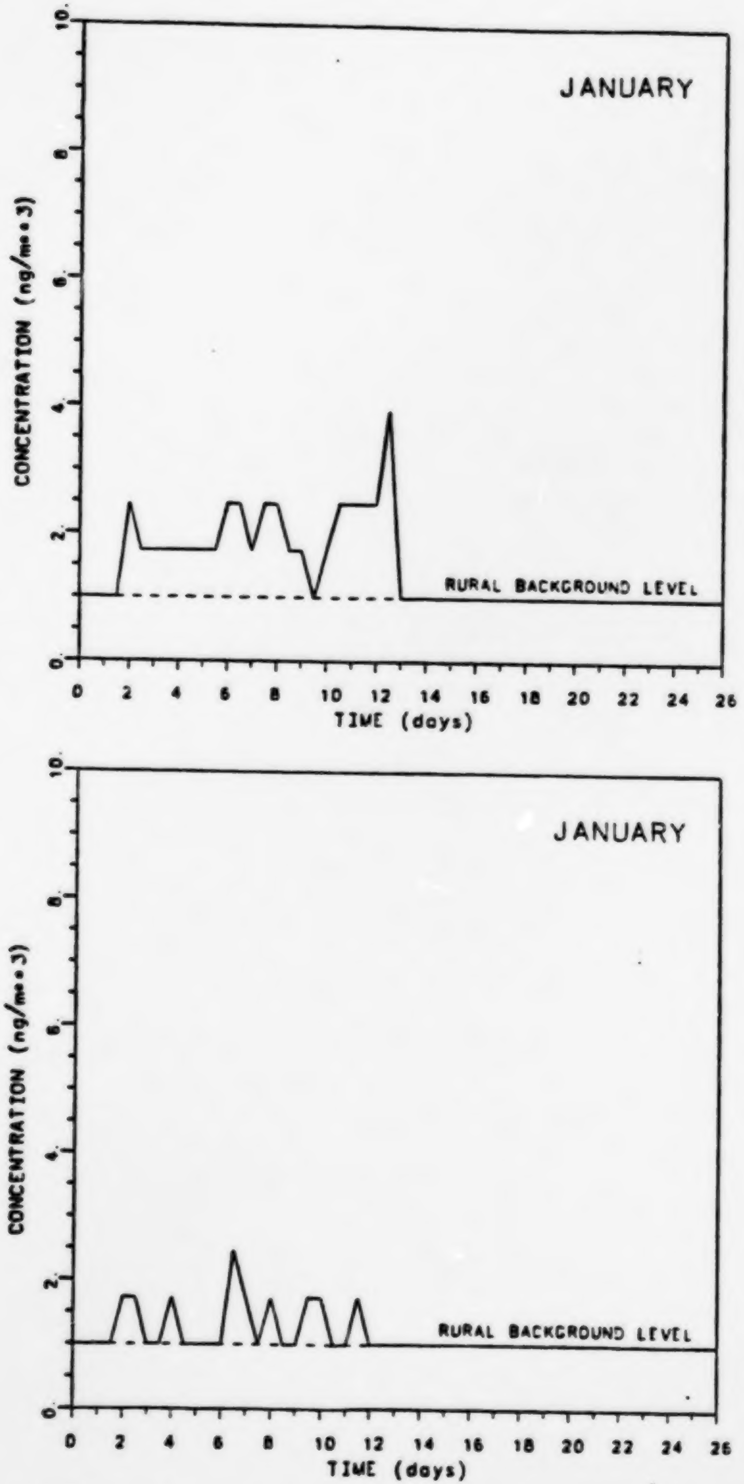


Exhibit G-19 Maximum PCB concentration time series at the coast in January with gaseous deposition velocities of (a) 0.2 cm/sec and (b) 0.5 cm/sec. Compare with Exhibit G-17a for 0.1 cm/sec.

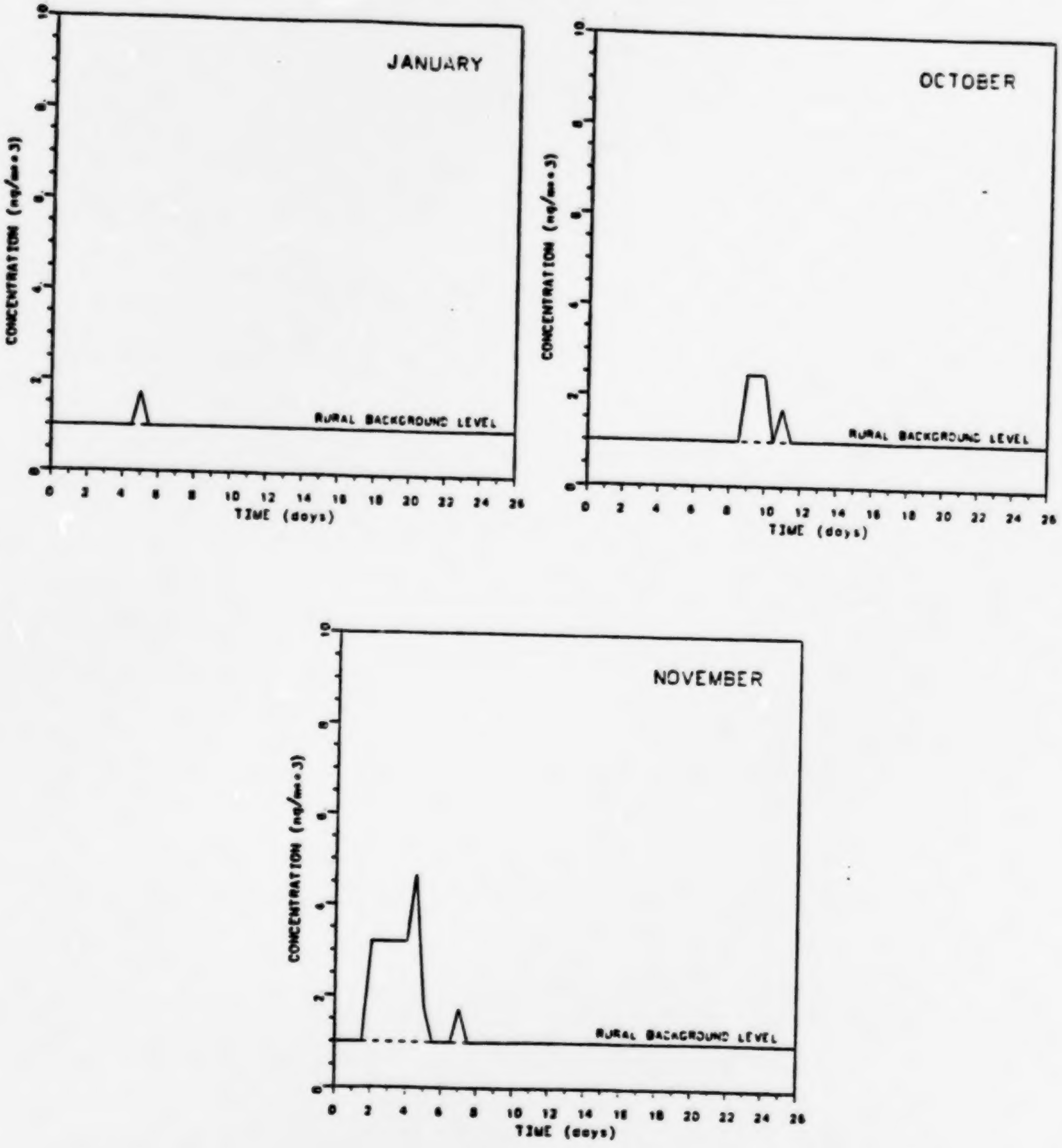


Exhibit G-20 PCB concentration time series at 24.5° north latitude, 97.8° west longitude for a) January, b) October, c) November.

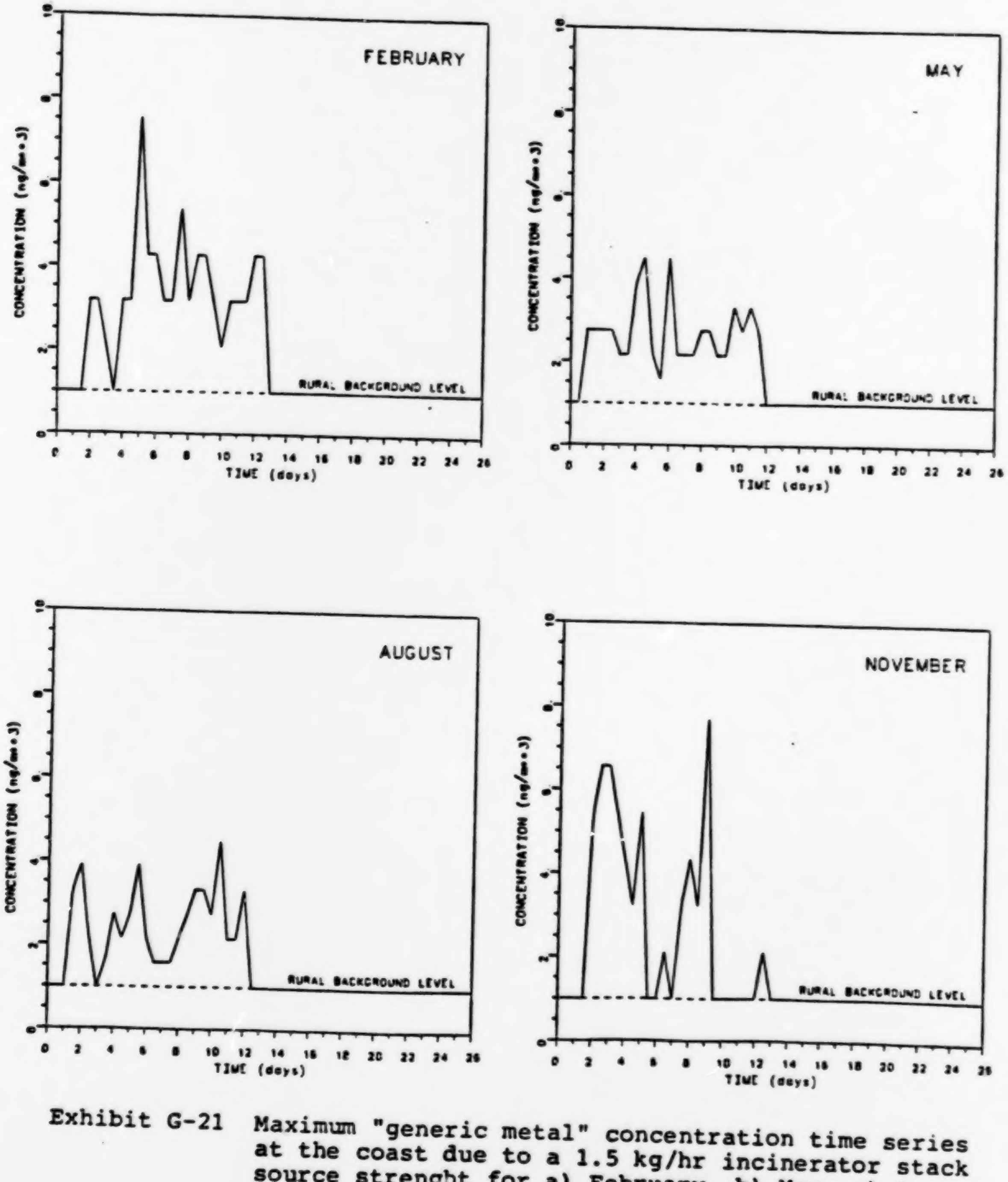


Exhibit G-21 Maximum "generic metal" concentration time series at the coast due to a 1.5 kg/hr incinerator stack source strenght for a) February, b) May, c) August d) November.

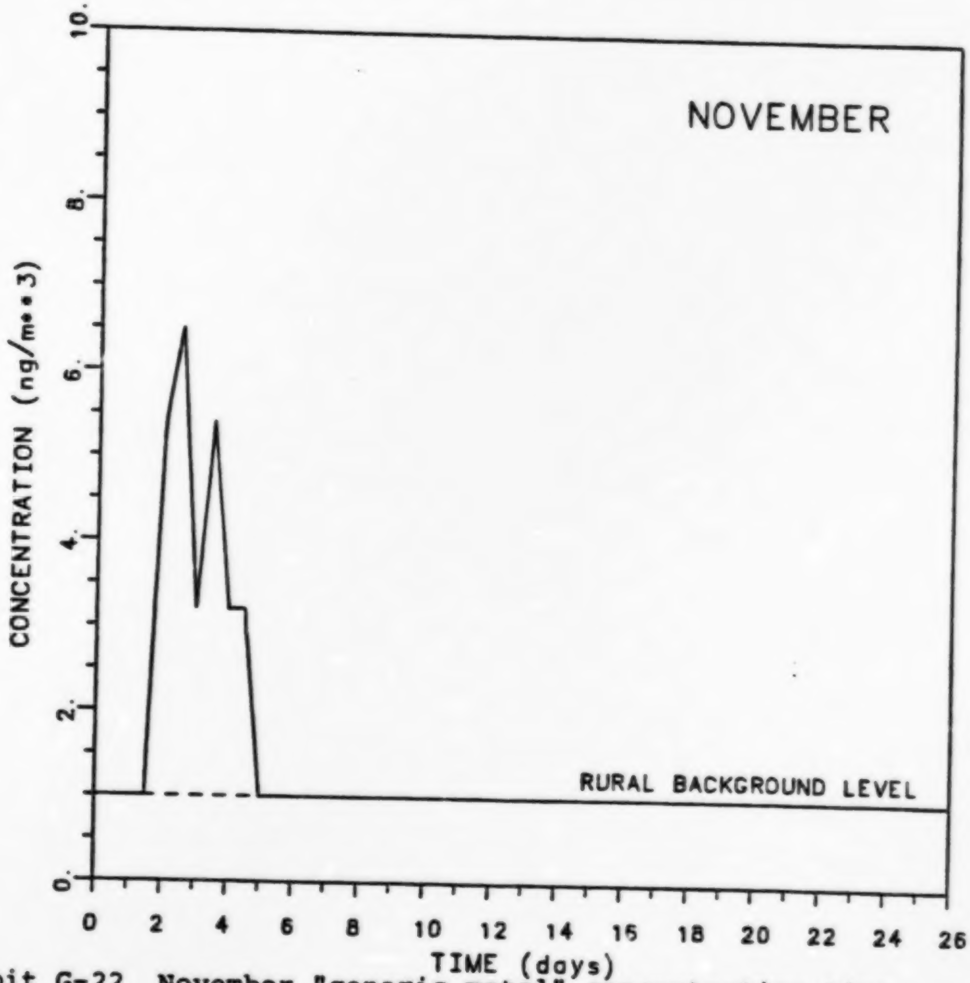


Exhibit G-22 November "generic metal" concentration time series at 24.5° north latitude, 97.8° west longitude.

Exhibit G-23

INTEGRATED EXPOSURES AND MAXIMUM DOSES BASED ON ONE KILOGRAM PER HOUR EMISSION RATE

	Integrated Exposure (ng-days/m3)	Maximum Dose (mg/kg/day)
Stack POHCs	11.0	6.38E-9
Stack PICs	11.0	6.38E-9
Metals	11.3	6.55E-9

Exhibit G-24

CALCULATION OF AVERAGE DAILY DOSE
FOR MOST EXPOSED INDIVIDUAL

$$\begin{array}{ccccc} \text{Unit Emission} & & \text{Unit Emission} & & \text{Dose} \\ \text{Average Daily} & & \text{Integrated} & & \text{Conversion} \\ \text{Dose} & = & \text{Exposure} & \times & \text{Factor} \\ \\ \dfrac{\text{mg}}{\text{Kg-day}} & & \dfrac{\text{ng-days}}{\text{m3}} & & \dfrac{\text{m3 - mg}}{\text{kg-ng-days-day}} \end{array}$$

$$\begin{array}{ccccccccccc} \text{Dose Conversion Factor} & = & \dfrac{22.8}{1} & \times & \dfrac{1}{70} & \times & \dfrac{1}{1{,}000{,}000} & \times & 0.65 & \times & \dfrac{1}{365} \\ \\ & & \dfrac{\text{m3}}{\text{day}} & & \dfrac{1}{\text{kg}} & & \dfrac{\text{mg}}{\text{ng}} & & & & \dfrac{1}{\text{days}} \\ \\ & = & 5.80\text{E-}10 \end{array}$$

Exhibit G-25

ACTUAL EMISSION RATES
(kg/hr)

	PCB Waste	EDC Waste
Stack POHCs	5.25E-3	1.25E0
Stack PICs	1.70E-9	7.53E0
Metals	6.0E0	1.00E+1
PIC Emissions Land-Based Units	5.10E-6	

Source: IEc Analysis

Exhibit G-26

CALCULATION OF MAXIMUM AVERAGE DAILY DOSE

$$\begin{array}{c}\text{Actual}\\ \text{Maximum Average}\\ \text{Daily Dose}\end{array} = \begin{array}{c}\text{1 kg/hr}\\ \text{Maximum Average}\\ \text{Daily Dose}\end{array} * \frac{\text{Actual Stack Emission Rate}}{\text{1 kg/hr}}$$

Source: IEc Analysis

Exhibit G-27

MAXIMUM AVERAGE DAILY DOSES
(mg/kg/day)

	PCB Waste	EDC Waste
Stack POHCs	3.35E-11	7.98E-9
Stack PICs	1.08E-17	4.80E-8
Metals	3.93E-8	6.55E-8
TOTAL	3.93E-8	1.21E-7

Source: IEc Analysis

Exhibit G-28

SENSITIVITY OF OCEAN INCINERATOR DOSE ESTIMATES TO ALTERNATIVE PIC GENERATION RATES FOR PCB WASTE

(mg/kg/day)

	Ocean-Based PIC Rate	Land-Based PIC Rate
Stack POHCs	3.35E-11	3.35E-11
Stack PICs	1.08E-17	3.25E-14
Metals	3.93E-8	3.93E-8
Total	3.93E-8	3.93E-8

Source: IEc Analysis

Exhibit G-29

CARCINOGEN
UNIT RISK FACTORS
(1/(mg/kg/day))

EDC	6.90E-2
PCB	4.34E0
Chloroform	7.00E-2
2,3,7,8 - TCDD	1.56E+5
Metals	1.62E+1

Source: Health Assessment Document for Polychlorinated Dibenzo-p-Dioxins (Review Draft), EPA, May 1984.

Exhibit G-30

MOST EXPOSED INDIVIDUAL RISKS

	PCB	EDC
Stack POHCs	1.45E-10	5.51E-10
Stack PICs	1.68E-12	3.36E-9
Metals	6.37E-7	1.06E-6
TOTAL	6.37E-7	1.06E-6

Exhibit G-31

SENSITIVITY OF OCEAN INCINERATION MOST EXPOSED INDIVIDUAL RISK ESTIMATES TO ALTERNATIVE PIC GENERATION RATES FOR THE PCB WASTE

	Ocean-Based PIC Rate	Land-Based PIC Rate
Stack POHCs	1.45E-10	1.45E-10
Stack PICs	1.68E-12	5.07E-9
Metals	6.37E-7	6.37E-7
Total	6.37E-7	6.37E-7

Exhibit G-32

SUMMARY OF LIFETIME RISK ESTIMATES FOR HIGHEST RISK POLLUTANTS (Average Lifetime Risks)

	NESHAPS	Ambient Air Data	35 County Study
Formaldehyde	7.0E-7	5.8E-5	1.5E-5
Benzene	9.8E-6	7.6E-5	2.7E-5
Chromium	1.0E-4	7.4E-5	2.0E-5
Cadmium	4.9E-6	4.2E-6	1.4E-6
Nickel	2.5E-5	4.9E-6	1.4E-6
Arsenic	1.4E-6	1.8E-5	1.4E-6
Trichloroethylene	2.8E-6	7.7E-6	1.1E-5
Perchloroethylene	7.0E-7	7.7E-6	9.8E-6
Ethylene dichloride	1.3E-5	N/A	2.1E-6
Ethylene oxide	1.5E-5	N/A	N/A
Carbon tetrachloride	4.2E-6	2.6E-5	2.8E-7
Ethylene dibromide	8.4E-6	N/A	1.4E-6
Chloroform	<7.0E-7	3.2E-5	1.4E-7
Gasoline vapors	N/A	N/A	1.1E-5
All Other	7.0E-6	9.8E-6	2.4E-5
PIC	2.5E-4*	2.4E-4*	2.2E-4
Radionuclides	5.6E-6*	5.6E-6*	5.6E-6*
Asbestos	3.5E-5*	3.5E-5*	3.5E-5*
Gasoline Marketing	1.4E-5*	1.4E-5*	
Total	5.0E-4	6.1E-4	3.9E-4

* Risk estimates from other EPA efforts.

Source: Preliminary Draft EPA staff report titled, The Magnitude and Nature of the Air Toxics Problem in the United States, E. Haemisegger, A. Jones, B. Steigerwald, V. Thomson, 1984.

Appendix H

ESTIMATION OF HUMAN EXPOSURES AND HEALTH RISKS RESULTING FROM MARINE SPILLS

This appendix describes the estimation of human health risks from exposure to PCB and EDC volatilizing from marine spills. Effects are estimated for hypothetical spills occurring in Mobile Harbor, at the burn site in the Gulf of Mexico, and at an intermediate location over the Continental Shelf off the Mississippi river delta. Specific locations for the burn zone and shelf spills are provided in Chapter 4 of the main report. Because volatilization from a spill would be a relatively short-lived phenomenon, our analysis only considers short-term acute health risks. In addition, we only estimate exposures which occur on land. Exposures to the ship's crew, workers participating in spill cleanups, and others on the water are not considered.

Three steps were used to estimate human health risks due to inhalation of pollutants volatilizing from a marine spill:

1. Estimation of volatilization rates for marine spills,
2. Calculation of pollutant concentrations at land locations downwind of the spill, and
3. Assessment of the human health impact of the resulting pollutant concentrations.

Each of these steps is discussed in a separate section of this appendix.

It is important to note that the analytic techniques and models used to estimate exposures and risks are not perfect predictors of risks which would result from an actual spill. Each of the steps in the estimation process is subject to uncertainty. Field verification studies of the estimated volatilization rates have not been conducted and monitoring studies of the air dispersion of pollutants have not been performed for the spill sites analyzed in this study. Further, the health impact of a spill is very sensitive to spill location and weather conditions at the time. A spill occurring in lower Mobile Harbor on a day when a strong wind is blowing from the north would be much less risky than a spill in the inner harbor area near Mobile on a day when light winds are out of the south. Our analysis considers only a small subset of the various locations and conditions under which a spill could occur. As a result, the estimates presented in this analysis should be viewed as rough characterizations of the health impacts which could result from marine spills of the PCB and EDC wastes.

ESTIMATION OF VOLATILIZATION RATES FROM MARINE SPILLS

To estimate volatilization rates from marine spills, one must determine whether the releases will be rapidly dissolved in the water or whether they will form a slick on the surface. Volatilization rates for slicks are estimated differently than rates for compounds dissolved in water. In our analysis, we assume that the PCBs would be mixed with oily compounds which would float on the surface and would behave like an oil spill. Volatilization rates from such a slick would be higher than the rates which would result from a PCB spill which dissolved or sank in the water. For the EDC wastes, we assume full dissolution in the water, given the highly soluble nature of EDC. However, if the EDC were dissolved in a substance with a specific gravity less than water and as a result formed a slick, volatilization rates would be higher than those used for this analysis.

To estimate volatilization rates from the PCB spill, we used the methodology described by Lyman in his analysis of volatilization rates for EDB in gasoline. The equation for calculating the percentage of a waste which will have volatilized from the water after t hours have elapsed is shown in Exhibit H-1. The values for the parameters used in the PCB volatilization rate calculations were provided by Lyman and Applied Science Associates, Inc. and are included in Exhibit H-2. For a more detailed discussion of the estimation procedure, see the paper by Lyman (1984), which is referenced in Exhibit H-1.

The procedure used to estimate volatilization rates from the EDC spill is based on the exponential decay function illustrated in Exhibit H-3. We used this equation to estimate volatilization at the ocean burn and continental shelf sites. To estimate volatilization in Mobile Harbor, the equation was modified by Applied Science Associates, Inc. to account for the flushing of pollutants from the harbor. This second equation is also provided in Exhibit H-3. The parameters used in the equations are provided in Exhibit H-4.

The results for the volatilization analysis are presented in Exhibits H-5 and H-6. Exhibit H-5 shows the percent of the pollutants which volatilize during the first day, the first five days, and the first ten days following the spill. Note that the rates for PCBs are the same regardless of the site. This occurs because we assume that the PCBs form a slick, and the volatilization rate from the slick is not affected by the depth of the water at the spill location. We assume other environmental parameters are the same for all three sites.

EDC is assumed to mix in the water column and thus the volatilization rate is affected by water depth at the spill location. Greater depths imply more mixing and lower volatilization rates. For the Mobile Harbor site, we assumed an average depth of 3 meters whereas for the ocean and continental shelf locations a mixing depth of 60 meters was used. As a result, volatilization rates for Mobile Harbor are higher than those for the ocean and continental shelf sites.

In general, Exhibit H-5 shows that the PCBs volatilize much less rapidly than EDC in Mobile Bay, on the Continental Shelf and at the burn site. Even after ten days, only 2.2 percent of the PCBs have volatilized from Mobile Harbor as opposed to 97 percent of the EDC. At the ocean and continental shelf sites, EDC volatilization is less rapid than PCB volatilization due to the greater mixing depth for the EDC. However, over the 10-day period, significantly more of the EDC spill will volatilize.

Exhibit H-6 presents the average emission rates (grams/-second) over the first day, the first five days, and the first ten days. For EDC, use of the one day averaging period results in estimates approximately three times higher than those for the ten day average for the Mobile Harbor site. For PCBs, the emission rate is not sensitive to the selection of an averaging period. The one day average is less than 1 percent higher than the ten day average. Because our health effect metrics are based on the values for short-term occupational exposures, we base our estimates of health risk on the one day averages, which result in the maximum concentrations to which people would be exposed.

ESTIMATING AMBIENT AIR CONCENTRATIONS AND HUMAN EXPOSURES

The second step in determining the human health consequences of marine spills of PCBs and EDC is to estimate the ambient air concentrations to which persons downwind of a spill would be exposed. To estimate the downwind concentrations for Mobile Harbor and for the continental shelf site, we used the virtual point source model for area sources described in EPA's Workbook of Atmospheric Dispersion Estimates (Turner 1970). In the virtual point source approach, an area source is modeled as a point source upwind of the actual area source. The virtual point source's distance upwind is that distance which would result in a plume width equivalent to the width of the area source at the actual location of the area source. Pollutant concentrations at groundlevel along the centerline of the plume were estimated using the gaussian approach described by Turner (page 6 of the Workbook).

For the Mobile Harbor site, we modeled two spill locations. As a worst case, we assumed the spill occurred one kilometer from the downtown Mobile waterfront. The second spill site was located in lower Mobile Bay fifteen kilometers from downtown Mobile. We estimate pollutant concentrations in downtown Mobile under both neutral (D stability) and stable (F stability) meteorological conditions. Pollutant concentrations under stable atmospheric conditions are worst case estimates. Neutral conditions represent the more typical situation. An average wind speed of three meters per second is used in the worst case analysis; six meters per second is used for the more typical case. In both cases we assume the wind is blowing the plume directly toward the city. Pasquill-Gifford dispersion coefficients were estimated for both analyses.

The area of the PCB spill was estimated to have a width of 1.34 kilometers, the maximum areal extent of a spill of 4000 metric tons before it begins to break up. For the EDC spill, the average width during the first day after the spill was estimated at 1650 meters based on the technique described by Csanady (Csanady, G., Turbulent Diffusion in the Environment, D. Reidel, London, 1973, p. 8). For this analysis, a dispersion coefficient of 10 meters per second was employed (personal communication, Dr. Mark Reed, Applied Science Associates, Inc.).

The results from the air dispersion modeling analysis for Mobile Harbor are included in Exhibit H-7. Concentrations of

PCBs in Mobile under stable atmospheric conditions ranged from 0.3 milligrams per cubic meter (mg/m3) for the spill occurring one kilometer from the city to 0.03 mg/m3 for the spill located fifteen kilometers out in the bay. Under neutral atmospheric conditions, concentrations are roughly fifty percent of those under stable conditions.

For the EDC spills, the concentrations under stable conditions are higher than those for PCBs and range from 229 mg/m3 for the one kilometer site to 19.7 mg/m3 for the fifteen kilometer site. Under neutral conditions, concentrations are lower by approximately a factor of ten. The primary reason for the higher concentrations of EDC is the more rapid volatilization of the compound from the water.

A similar methodology was used to estimate concentrations downwind of the continental shelf site. However, the continental shelf spill was assumed to occur 75 kilometers off-shore. Emission rates for this case are the one day averages shown in Exhibit H-6.

The results of the continental shelf site dispersion analysis are presented in Exhibit H-8. For both PCBs and EDC these concentrations are lower than those estimated for the harbor site located fifteen kilometers from Mobile. These estimates assume no pollutant decay or deposition; over the longer transport distances from the continental shelf site these processes would probably further lower the concentrations relative to those for the Mobile sites.

For spills occurring at the ocean burn site, we used the dispersion modeling results presented in Appendix G. We scaled the estimate of the maximum concentration (approximately 6.0 nanograms per cubic meter) which would occur during the year at a coastal site given one kilogram per hour of emissions to reflect actual emissions from the spill area. The emission rates for the PCB and EDC wastes at this site are included in Exhibit H-6.

The results of the dispersion analysis for the ocean burn site are presented in Exhibit H-9. Because this analysis is based on the maximum annual concentrations observed along the Gulf Coast, these concentrations represent worst case estimates. For the PCB spill, maximum ambient air concentrations were estimated to be 0.0008 mg/m3. For EDC, concentrations of 0.006 mg/m3 would be observed. As one might expect given the increased distance from the spill to the shoreline, the concentrations resulting from spills at the ocean site are even lower than those for the continental shelf site.

ASSESSMENT OF HUMAN HEALTH IMPACTS

To assess the possible human health effects associated with pollutant spills at the Mobile Harbor, Continental Shelf, and ocean burn sites, we compared the estimates of ambient pollutant concentrations described above with maximum concentration limits recommended for occupational exposures. Pollutant concentrations are compared to both short and long term exposure limits. The short term exposure limit (STEL) for a chemical is the concentration to which workers can be exposed continuously for a short period of time (15 minutes) without suffering irritation, chronic or irreversible tissue damage, or narcosis of a sufficient degree to increase the likelihood of an accident. The longer term exposure limit is referred to as the threshold limit value (TLV) and represents the concentration to which workers may be repeatedly exposed without suffering adverse effects. The TLV is based on exposures occuring eight hours per day five days per week. Because exposures from a spill would occur twenty-four hours per day, and potentially for more than five days, we have adjusted the TLVs to represent acceptable air concentrations based on continuous weekly exposures. This adjusted TLV is the pollutant concentration which, if breathed continuously throughout the week, would result in a weekly dose equivalent to the dose received by a worker exposed to the compound eight hours per day, five days per week. The adjusted TLV and STEL values are included in Exhibit H-10.

In Exhibit H-11, we compare the ambient pollutant concentrations resulting from spills at the three sites (Exhibits H-7 through H-9) with the STELs for the two chemicals. The values in this table represent the ratio of the estimated pollutant concentration to the STEL value. When this value exceeds one, the STEL is exceeded. The data in Exhibit H-11 suggest that the STEL for PCBs is never exceeded for the spills occuring at either of the Mobile Harbor locations. For spills of EDC the STEL is exceeded only under the stable atmospheric conditions and only at the 1 kilometer from the Mobile Harbor location. Thus, the analysis indicates that for EDC spills occurring near Mobile, acute health effects could result under stable atmospheric conditions.

Exhibit H-11 also contains the pollutant concentration to STEL ratios for the ocean burn, and continental shelf sites. Spills occurring at these locations will result in human exposures to pollutant concentrations at least 50 times lower than the STEL.

Since exposures to chemicals volitalizing from spills would likely occur for periods longer than the fifteen minutes on which the STEL is based, it is also useful to compare pollutant concentrations with the TLV values. These are more representative of the safe limits for longer term exposures to these chemicals. For Exhibit H-12 we have calculated the ratio of the pollutant concentration to the TLV. At the 1 kilometer from Mobile site, the TLVs are exceeded for both pollutants. At the 15 kilometer location, the EDC TLV is exceeded under stable atmospheric conditions. Under neutral conditions, the EDC concentrations are below the TLV. For the PCB spills at the 15 kilometer site, air concentrations are below the TLV for all atmospheric conditions. This analysis suggests that spills of either chemical at the 1 kilometer site can pose important health risks regardless of the meteorological conditions. For spills at the 15 kilometer sites, under the worst case atmospheric conditions, only the EDC wastes results in significant health risks.

Exhibit H-12 also contains the pollutant concentration to TLV ratios for the continental shelf and ocean burn site. For EDC and PCB spills at the continental shelf and at the ocean burn site, the TLV is never exceeded along the Gulf coast.

Exhibit H-1

Estimation of Volatilization Rates For PCB

% Volatilized = $(1 - n/n_o) \times 100$

$$n/n_o = \exp(-k_G A P \gamma t / n_T)$$

where n = number of moles of PCB present in spill at time t

n_o = number of moles of PCB present in spill initially

n_T = total number of moles present in spill (PCB plus oil)

t = time (h)

A = spill surface area (m^2)

P = vapor pressure of pure PCB (atm)

γ = PCB activity coefficient

$$k_G = 0.0292\, u^{0.78}\, d^{-0.11}\, S_c^{-0.67}/RT$$

where u = wind speed (m/hr)

d = equivalent spill diameter

S_c = gas-phase Schmid number (dimensionless)

R = gas constant (8.21×10^{-5} atm m^3/mol K)

T = temperature (K)

Source: Lyman, W. 1984. Volatilization of EDB From EDB - Containing Gasoline Following Small Spills on Human Skin. A.D. Little, Inc. Cambridge, MA.

Exhibit H-2

Values of Parameters Used To Estimate PCB Volatilization Rate

	Value
Surface Area (A)	1.8 km^2
Vapor Pressure (P)	1.01 E-7 atm
Activity Coefficient (γ)	10*
Molecular Weight	328.4
Wind Speed (u)	21,600 m/hr.
Spill Diameter (d)	1650 m
Schmidt no. (S_c)	3*
Temperature (T)	298°(k)

---

* Based on personal communication with W. Lyman.

Exhibit H-3

Estimation of Volatilization Rates For EDC

% volatilizing after Time t = $(1 - c/c^{o}) \times 100$

(1) For Ocean and Continental Shelf Sites

$$c/c^{o} = \exp(-kt/z)$$

Where:

c_o = Initial concentration of pollutant

c = Concentration of pollutant at time t.

k = Volatilization rate constant

t = time (hr)

z = Depth of mixing zone (cm)

(2) For Mobile Bay

$$c/c_o = \exp(-kt/z - k_2t)$$

Where:

k_2 = Rate constant for flushing of pollutants from bay.

Source: Lyman, W., Roehl, W. and Rosenblatt, D. 1982. Handbook of Chemical Property Estimation Methods. McGraw Hill, NYC.

Exhibit H-4

Values of Parameters Used to Estimate Volatilization Rates For EDC

	Ocean/Continental Shelf Site	Mobile Bay
Volatilization Rate Constant (k)	3 cm/hr	3 cm/hr
Flushing Rate Constant (k_2)		0.005 hr
Depth of Mixing Zone (z)	6000 cm	300 cm

Source: Personal communication, Dr. Mark Reed, Applied Science Associates.

Exhibit H-5

POLLUTANT VOLATILIZATION RATES

	Percent Volatilizing From					
	Mobile Bay		Continental Shelf		Burn Site	
Elapsed Time	PCB	EDC	PCB	EDC	PCB	EDC
1 day	0.2	30.2	0.2	1.2	0.2	1.2
5 days	1.1	83.5	1.1	5.8	1.1	5.8
10 days	2.2	97.3	2.2	11.3	2.2	11.3

Source: IEc Analysis

Exhibit H-6

AVERAGE EMISSION RATES
(g/sec)

Emission Rate Average Over	Mobile Bay		Continental Shelf		Ocean Burn Site	
	PCB	EDC	PCB	EDC	PCB	EDC
1 day	35.5	6647	35.5	264	35.5	264
5 days	35.3	3671	35.3	255	35.3	255
10 days	35.2	2139	35.2	249	35.2	249

Source: IEc Analysis

Exhibit H-7

DISPERSION ANALYSIS FOR MOBILE BAY SPILLS

Spill Location	Maximum 1 Day Concentrations (mg/m3)	
	PCB Waste	EDC Waste
1 kilometer from Mobile		
Neutral Conditions	0.2	23.7
Stable Conditions	0.3	229.0
15 kilometers from Mobile		
Neutral Conditions	0.01	1.5
Stable Conditions	0.03	19.7

Source: IEc Analysis

Exhibit H-8

DISPERSION ANALYSIS - CONTINENTAL SHELL SPILLS
(mg/m3)

	PCB Waste	EDC Waste
Neutral Conditions	0.003	0.020
Stable Conditions	0.021	0.210

Source: IEc Analysis

Exhibit H-9

DISPERSION ANALYSIS - OCEAN BURN SITE SPILLS
(mg/m3)

	PCB Waste	EDC Waste
Neutral Conditions	0.0008	0.006

Source: IEc Analysis

Exhibit H-10

EXPOSURE LIMITS FOR PCB AND EDC

	Threshold Limit Value*	Short-Term Exposure Limit
PCB	0.16	1.0
EDC	12.5	60.0

---

* Adjusted to reflect 24hr/day 7day/wk exposures

Source: *Threshold Limit Values for Chemical Substances and Physical Agents in the Work Environment With Intended Changes for 1983-84*, American Conference of Government Industrial Hygienists, Cincinnati, OH, 1983.

Exhibit H-11

POLLUTANT CONCENTRATIONS/SHORT TERM EXPOSURE LIMIT

	Mobile Bay		Continental Shelf	Ocean Burn Site
	1k	15k		
PCB				
Neutral Conditions	0.2	1.0E-2	3.0E-3	7.8E-4
Stable Conditions	0.3	3.0E-2	2.1E-2	
EDC				
Neutral Conditions	4.0E-1	2.5E-2	3.3E-4	9.3E-5
Stable Conditions	3.8	3.3E-1	3.5E-3	

Source: IEc Analysis

Exhibit H-12

POLLUTANT CONCENTRATION/THRESHOLD LIMIT VALUE

	Mobile Bay		Continental Shelf	Ocean Burn Site
	1k	15k		
PCB				
Neutral Conditions	1.3	6.3E-2	1.9E-2	1.89E-3
Stable Conditions	1.9	1.9E-1	1.3E-1	
EDC				
Neutral Conditions	1.9	1.2E-1	1.6E-3	4.47E-4
Stable Conditions	18.3	1.6	1.7E-2	

Source: IEc Analysis

APPENDIX I. ECOSYSTEM EFFECTS DUE TO MARINE RELEASES

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1. Overview

An ocean risk assessment model system (Reed et al. 1984a, b, 1983a) has been used to estimate the effects of accidental and intentional releases of toxic wastes or residues on the biota of Mobile Bay and the Gulf of Mexico. As shown in Figure 1-1, the model system consists of water column, sediment, and ecosystems submodels, coupled to simulate specific waste disposal scenarios.

The water column pollutant transport model uses seasonal or annual estimates of upper and lower water column mean velocity fields and water column chemistry to estimate the hydrodynamic transport of pollutants. Ultimate loss of pollutants from this model is either to the sea floor, out the open boundaries, or via decay processes. The sediment model describes pollutant incorporation into and release from the sediments as a function of time. The ecosystems model is spatially averaged, and includes formulations of physiological and metabolic processes. The ecosystems model receives input from the water column and sediment models in the form of pollutant concentrations. These environmental concentrations determine the biological effects. Bioaccumulation is computed as a function of diet throughout the food web. Each of these model system components will be described in more detail in the following sections.

The model system has been used to estimate the ecosystem impacts of accidental spills of polychlorinated biphenyls (PCBs) and ethylene dichloride (EDC) in Mobile Bay, on the continental shelf, and in the offshore burn zone. For the PCB waste, we assume a 4000 metric ton (mt) load consisting of 35% Auroclor

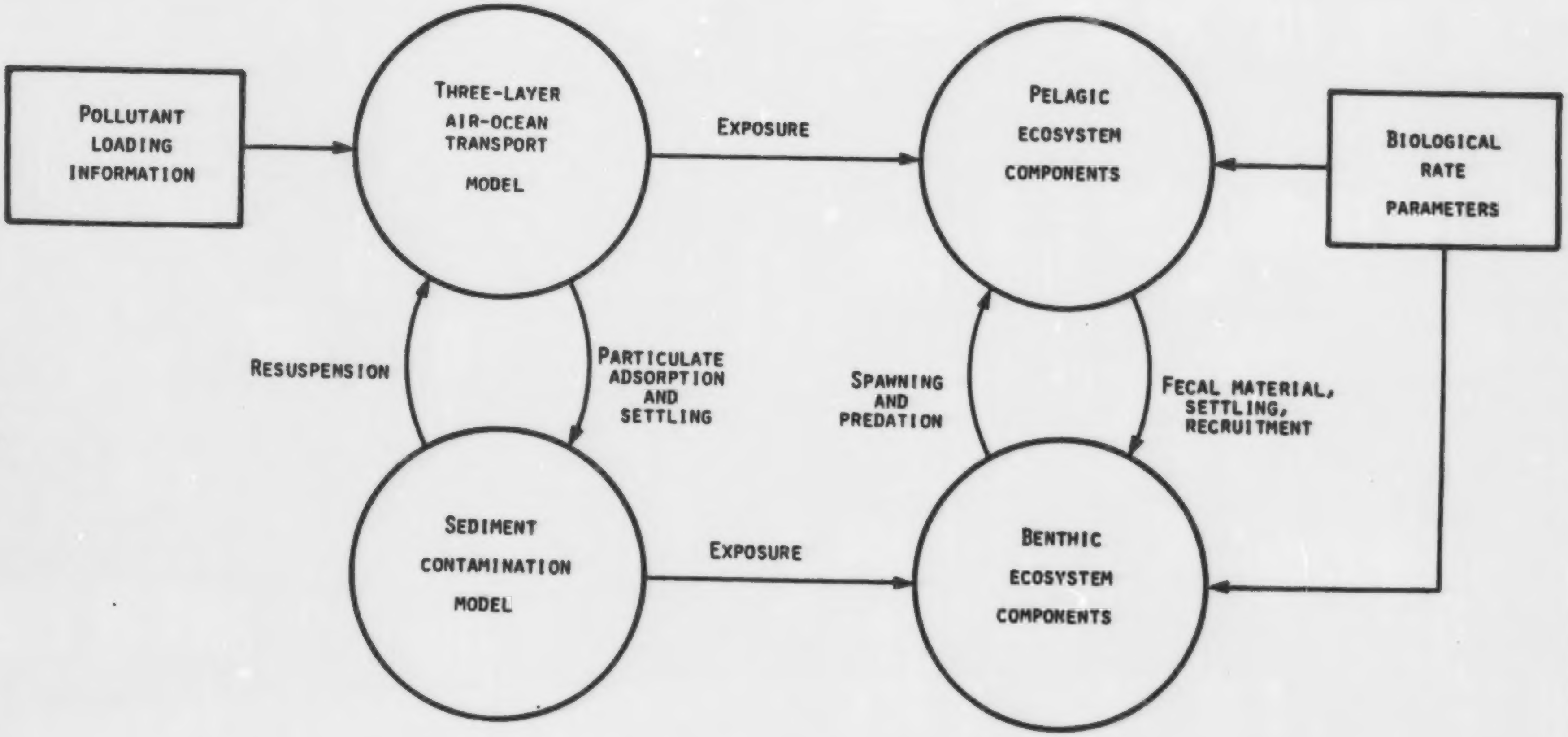
POLLUTANT LOADING INFORMATION
THREE-LAYER AIR-OCEAN TRANSPORT MODEL
EXPOSURE
PELAGIC ECOSYSTEM COMPONENTS
BIOLOGICAL RATE PARAMETERS
RESUSPENSION
PARTICULATE ADSORPTION AND SETTLING
SPAWNING AND PREDATION
FECAL MATERIAL, SETTLING, RECRUITMENT
SEDIMENT CONTAMINATION MODEL
EXPOSURE
BENTHIC ECOSYSTEM COMPONENTS

Figure 1-1. Model system interaction schematic.

1254, and 65% of some other non-hazardous substance (e.g. light fuel oil). For the EDC waste, the total load is 3800 mt, consisting of 50% EDC. An atmospheric transport model (Appendix G) has been coupled to the ocean risk assessment model system to estimate impacts of PCBs and heavy metals deposited to the sea surface from the incinerator stack plume. The atmospheric model includes temporal and spatial variations in winds, and both dry and wet deposition.

Study Strengths and Weaknesses

The model system applied here is a predictive tool, used to quantitatively estimate possible effects before a proposed action is initiated. The major strength of the modeling approach used here is that it forces explicit recognition of assumptions, and unifies what is known about relevant physical, biological, and chemical processes into an internally consistent mathematical framework, such that quantitative impact estimates are produced. This approach then allows a decision-maker to investigate the effect of alternate assumptions and project plans on potential impacts. Such complex models entail considerable uncertainty, however, and their application should be accompanied by sensitivity and verification studies whenever possible. Only limited sensitivity studies have been performed for the applications reported here. The uncertainties associated with these impact estimates have not been evaluated directly, although these results can be taken as "best estimates" given the model assumptions and limitations discussed here.

Processes in the full model system include:

(1) spatially variable two layer hydrodynamic transport,

(2) particulate/dissolved phase partitioning in the water column and sediments,

(3) pollutant sequestering from the water column to the sediments via particulate adsorption and settling,

(4) stochastic sediment resuspension,

(5) bioaccumulation from the environment (water and/or sediments) and food web dynamics,

(6) biomass (and fisheries yield) reductions through growth, reproductive rate reductions, or mortality rate increase.

System and study weaknesses are as follows:

(1) The microlayer has not been incorporated explicitly into the ecosystems model. Since microlayer concentrations of PCB and other pollutants have been recorded at levels exceeding underlying water concentrations by factors of 10 to 10^6, the incorporation of linkages between the microlayer and pelagic ecosystem may be desirable in the future.

(2) Time limitations have prevented application of the standard pollutant transport model in Mobile Bay. Thus horizontal geographical distribution of pollutants, incorporation of PCB into the sediment, and adsorption/dissolution partitioning in the water column have been estimated, rather than rigorously modeled. Volatilization of both PCB and EDC has been neglected. Although consistent with a "worst case" analysis, the impact of a PCB spill on ecosystem dynamics has probably been over-estimated. More realistic effects estimates could be obtained for this area.

(3) An important ecosystem impact parameter is recovery time following an accidental spill. Such recovery is largely a function of biological immigration from surrounding non-impacted areas, a process not modeled here. Establishment of a series of adjacent ecosystem models with the ability to transfer biomass from one to another as a function of organism mobility and hydrodynamic transport will allow much better estimates of recovery time than those given here.

(4) The hydrodynamics underlying the continental shelf and incineration zone pollutant transport simulations represent smoothed annual average current fields from a compendium of sources. Although the general oceanographic features of the Gulf of Mexico appear to be well represented, specific details, notably in shallower nearshore regions, appear questionable in many areas. Effects of seasonal variability have also been eliminated. A unified hydrodynamic modeling effort would place the transport and impact estimates on a firmer scientific footing.

(5) A metals activity model component will be necessary to permit quantified impact estimates of metals on the ecosystem. The quantification of such impacts is important for the microlayer under normal incineration operations, and for the entire ecosystem, particularly the benthos, in accidental spill situations.

(6) Mortality rates are computed based on total PCB concentrations in water and sediments. Although consistent with most bioassay procedures for measuring exposure, most bioassay work is performed in relatively particulate-free systems. The presence of particulate matter in the oceans plays an important role in sequestering toxic organics. The ecosystem model as applied here therefore probably overestimates mortality rates, especially at low levels of exposure.

2. Ocean Transport Model

The transport model computes the movement of pollutants in the water column. Empirical or modeled measures of mean seasonal currents provide the basis for transport estimates. Since mean surface and bottom currents may be quite different, a two-layered representation is used. Transport calculations are performed on a spherical coordinate grid system. This grid system takes into account the curvature of the earth's surface, an important consideration when computing transport over distances exceeding a few hundred kilometers. The size of the grid cells used defines the limit of the model to represent small scale processes. In general, the use of smaller grid cells will improve the model's capability to realistically represent the basic physics of pollutant transport, subject to limitations in data resolution. The use of smaller grid cells necessitates the use of more cells to cover a given region. The computational time and cost to operate the model is directly proportional to the number of grid cells, so these factors serve as impetus to keep grid cells large. In any particular application, a balance must be achieved between the need to resolve small scale processes on the one hand, and the need to control simulation costs on the other. This balance is achieved more easily as experience is gained in model application.

The pollutant transport model system can be applied in either a single or multi-layered mode. When the single-layer mode is used, vertically averaged currents are assumed to be acting throughout the entire water column. The multi-layer mode uses both near-surface and deeper currents. The surface currents act to a specified depth, generally to agree with the seasonal/annual pycnocline. (The pycnocline marks a depth at which a relatively sudden change in water density occurs. Vertical density changes of this sort are important because they tend to inhibit vertical mixing processes, and may therefore cause higher pollutant concentrations in surface waters than might otherwise be expected.) Bottom currents govern at depths below the pycnocline. Since mean bottom currents are often an order of magnitude slower than surface currents, use of the multi-layer model generally provides a more realistic assessment of pollutant transport.

2.1 Data Sources and Velocity Field Specifications

The basic data source for surface currents is the series of atlases of ship drift observations produced by the Naval Oceanographic Office (1977). Deep ocean currents for the present study have been derived from a three-dimensional hydrodynamic model of the Gulf of Mexico (Blumberg and Mellor, 1979).

The model grid for this study is 0.25 degrees in resolution. Once this grid was established relative to land, open ocean boundaries and bottom topography, a mean velocity vector was determined by interpolation for each grid cell. This interpolation was performed subject to horizontal mass conservation constraints. Figure 2.1-1 shows the boundaries of the transport grid as well as bathymetric contours used for the

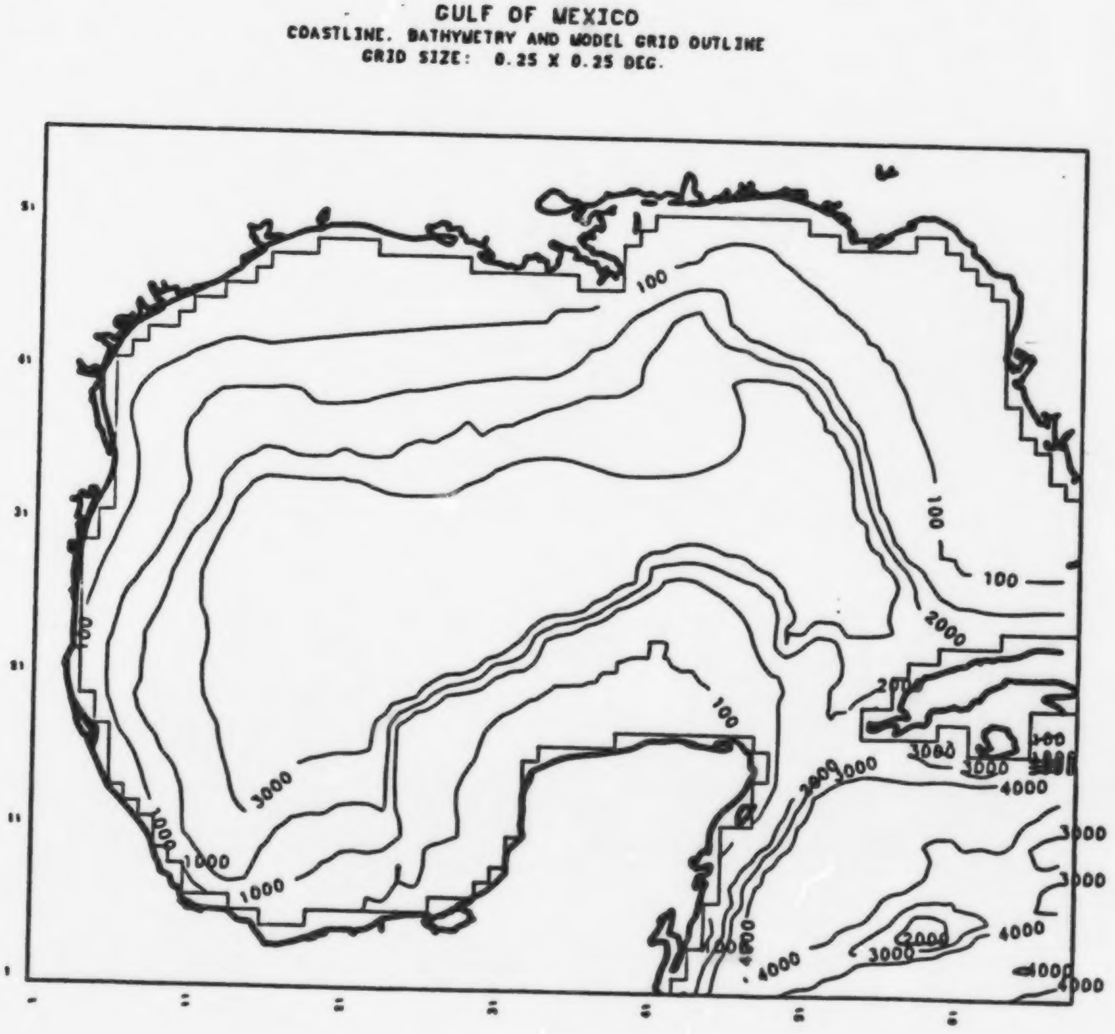
GULF OF MEXICO
COASTLINE, BATHYMETRY AND MODEL GRID OUTLINE
GRID SIZE: 0.25 X 0.25 DEG.
100
1000
2000
3000
4000

Figure 2.1-1. Model grid and bathymetry of the Gulf of Mexico.

Gulf of Mexico. Figures 2.1-2 and 2.1-3 show the associated annual average surface and deep ocean velocity fields.

2.2 Water Column Model Formulations

A majority of the man-made pollutants of concern in the marine environment, including hydrocarbons, heavy metals, radionuclides, chlorinated hydrocarbons and other organic compounds, are readily adsorbed onto suspended particulate matter and may eventually settle to the sea floor. Adsorption - desorption and settling dynamics are therefore included in the model system. This is necessary to properly simulate pollutant transport and fate, and thereby to estimate pollutant concentrations within biological trophic compartments. Food may be a more important factor in bioconcentration calculations than water (Fowler 1982; O'Connor and Pizza 1983), and it is therefore important to provide reasonable estimates of biologically available concentrations at the primary and secondary production levels, to achieve correct estimates of concentrations at the higher trophic levels.

A Lagrangian marker particle approximation to the advection-diffusion equation, following the development of Lange (1973), Spaulding (1976), and Reed (1980), is used to predict pollutant concentrations as a function of space and time. This numerical methodology is particularly useful in the control of numerical diffusion errors and in facilitation of settling computations. A waste dumping scenario at a specific location is simulated by releasing one or more particles at discrete time

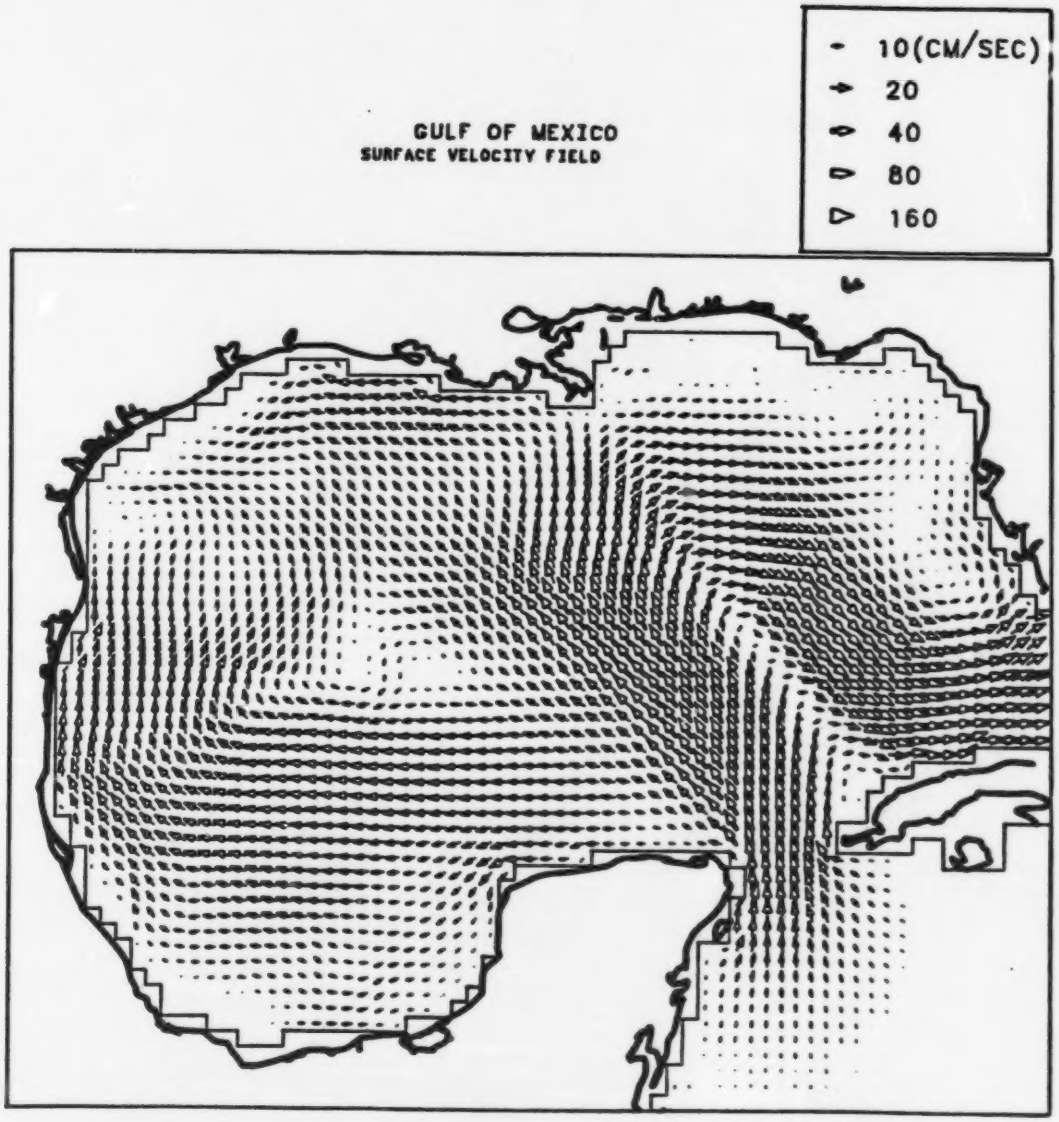

Figure 2.1-2. Surface velocity field for Gulf of Mexico.

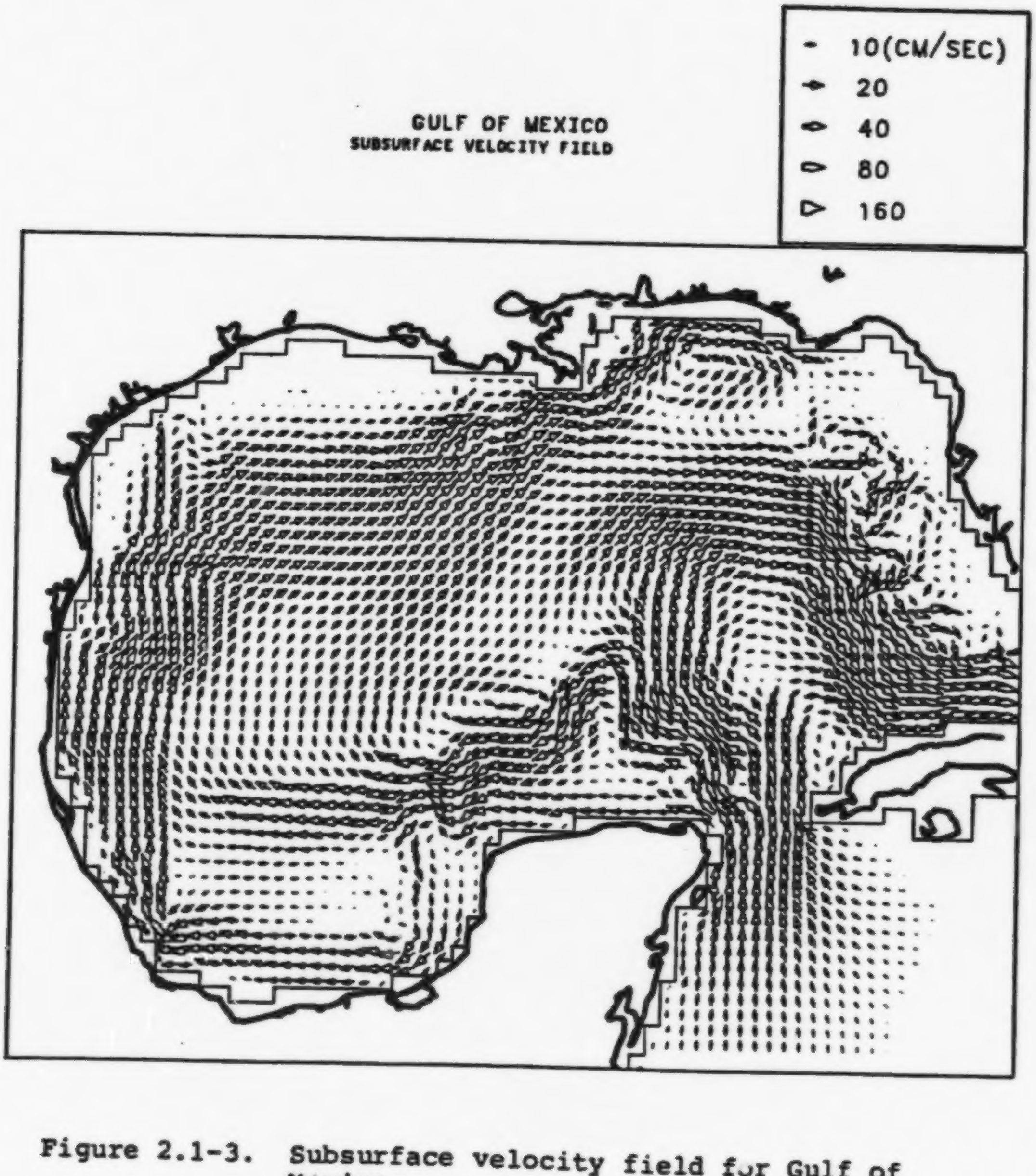

Figure 2.1-3. Subsurface velocity field for Gulf of Mexico.

intervals. Each particle represents a specific mass of pollutant, and particles are assigned settling velocities according to a specified probability distribution. Dissolved phase pollutants are associated with neutrally buoyant particles. Each particle carries a time tag, so that pollutant decay can be calculated analytically. A large number of particles, on the order of 2000 per simulation, is necessary to supply reasonable estimates of the dispersion process. If too few particles are used, the concentration distributions appear erratic and discontinuous. Greatly increasing the number of particles increases computational costs with little or no gain in simulation quality.

Total pollutant concentrations are resolved into adsorbed and dissolved phases assuming equilibrium linear partitioning (e.g. Bierman and Swain, 1982):

$$f = 1/(K_p C_{ss} + 1)$$

in which f is the fraction of pollutant in dissolved form, K_p is a dimensionless partition coefficient and C_{ss} is the suspended solids concentration. The partition coefficient is often available from experimental data. Otherwise, from Voice et al. (1983), we estimate K_p by

$$\log K_p = 0.748 \log K_{ow} - 0.648 \log C_{ss} - 0.131 \log f_{oc} + 0.364$$

where K_{ow} is the octanol-water partition coefficient, C_{ss} is suspended solids concentration in mg/l, and f_{oc} is the weight fraction of solids comprised of organic carbon. The particulate adsorption and settling processes in the model system result in a long term flux of pollutants from the water column to the sea floor.

Once the ratio of dissolved and adsorbed pollutant has been determined, the adsorbed fraction is allocated among the distribution of settling velocities for natural particles and aggregates (e.g. Hawley (1982), or Carder et al. (1982)). This allocation can be uniform, or weighted by particle size if desired. The adsorbed fraction which adheres to non-settling particles and the dissolved fraction are treated the same during transport computation. Movement of these particles results primarily from horizontal advection with random dispersion in both the horizontal and vertical planes. Pollutants which adhere to particles with non-zero settling velocities descend at the associated settling velocity modified by a random diffusion term (Reed, 1980).

2.3 Sediment Model Formulations

Eventually each pollutant particle encounters one of three fates: 1) it settles to the ocean floor, 2) it is transported outside the model domain, or 3) it decays to background levels.

(Volatilization has not yet been included in model capabilities.) Through resuspension, particles which settle to the ocean floor in depths of 100 m or less may be re-introduced to the water column. Resuspension results from wave and storm-generated turbulence which is governed by wave height and period frequency distributions. By overlaying the Summary of Synoptic Meteorological Observations (SSMO) table "Percent of Frequency of Wave Height vs. Wave Period" for a desired area with a "Wave Induced Sediment Suspension" table (Reed et al. 1983), the frequency with which sediments are resuspended at various depths can be determined. A random number generator is invoked to initiate resuspension at stochastic intervals.

The governing differential equation for vertical pollutant transport within the sediments, inclusive of a first order decay term is:

$$\frac{\partial C}{\partial t} = D \frac{\partial^2 C}{\partial z^2} - kC \tag{2.3-1}$$

in which C is pollutant concentration, D is a dispersion coefficient which includes bioturbation effects, t is time, z is distance from the water-sediment interface (positive downward) into the sediments, and k is the pollutant decay rate. The term on the left hand side of the above equation is the rate of change of pollutant concentration with time. The first term on the right hand side represents the vertical dispersion of the pollutant due to biological and wave-induced sediment motion. The second term on the right accounts for pollutant decay with time.

2.4 References

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3. Ecosystem Model

3.1 Introduction

Ideally, a spatially detailed ecological model of Mobile Bay, the Gulf Coast continental shelf, and the offshore burn site would be desirable. However, such models are computationally cumbersome and require more data and development time than are presently available (Reed et al. 1984). Therefore, a general ecosystem model was developed and applied to three representative sites which could be impacted by a toxic waste spill, or the plume from normal operational incineration. The first site is Mobile Bay, where wastes will be loaded. A second version of the ecosystems model represents the continental shelf enroute to the incineration site. The third reflects the ecosystem at the offshore incineration site (26°20'N - 27°00'N and 93°20'W - 94°00'W).

3.2 Ecosystem Model Structure and Dynamics

The organisms are grouped into trophic compartments which form a simplified food web. Table 3-1 summarizes the organisms included in and typical of the compartments, and Figure 3-1 diagrams energy (carbon) flow in the system. Vertical structure is represented in the model by a two-layered water column, a surface-mixed layer considered equivalent to the euphotic zone, and a bottom layer, below which are the sediments. Phytoplankton and the pelagic food chain (zooplankton and pelagic fishes, #5-8) are assumed to exist only in the surface layer, and benthic and

Table 3-1. Compartments included in the ecosystem model by site location: M = Mobile Bay, S = shelf (enroute), O = offshore incineration site, + = present in model run, 0 = biomass assumed zero.

#	Compartment Name	Organisms Included	M	S	O
1	phytoplankton	diatoms, flagellates	+	+	+
2	benthic flora	diatoms	+	0	0
3	benthic deposit feeders	nematodes, polychaetes crustaceans, mollusks echinodems, (most infauna)	+	+	+
4	demersal detritivores	decapods including shrimp, lobsters, crabs	+	+	0
5	herbivorous zooplankton	crustaceans (mainly copepods) salps, appendicularians, invertebrate larvae (i.e. of #3,4,5,6,9)	+	+	+
6	carnivorous zoolplankton	crustaceans, chaetognaths, ctenophores,jelly fish, squid, predatory crabs, icthyoplankton	+	+	+
7	small pelagic fishes	menhaden, herrings (planktivorous fishes)	+	+	+
8	large pelagic fishes	tunas, sharks (predatory fishes)	0	+	+
9	benthic invertebrate predators	mollusks, asteriods, corals (suspension feeders and predators)	+	+	+
10	demersal fishes	gadoids, flatfish deep-sea demeral species	+	+	+

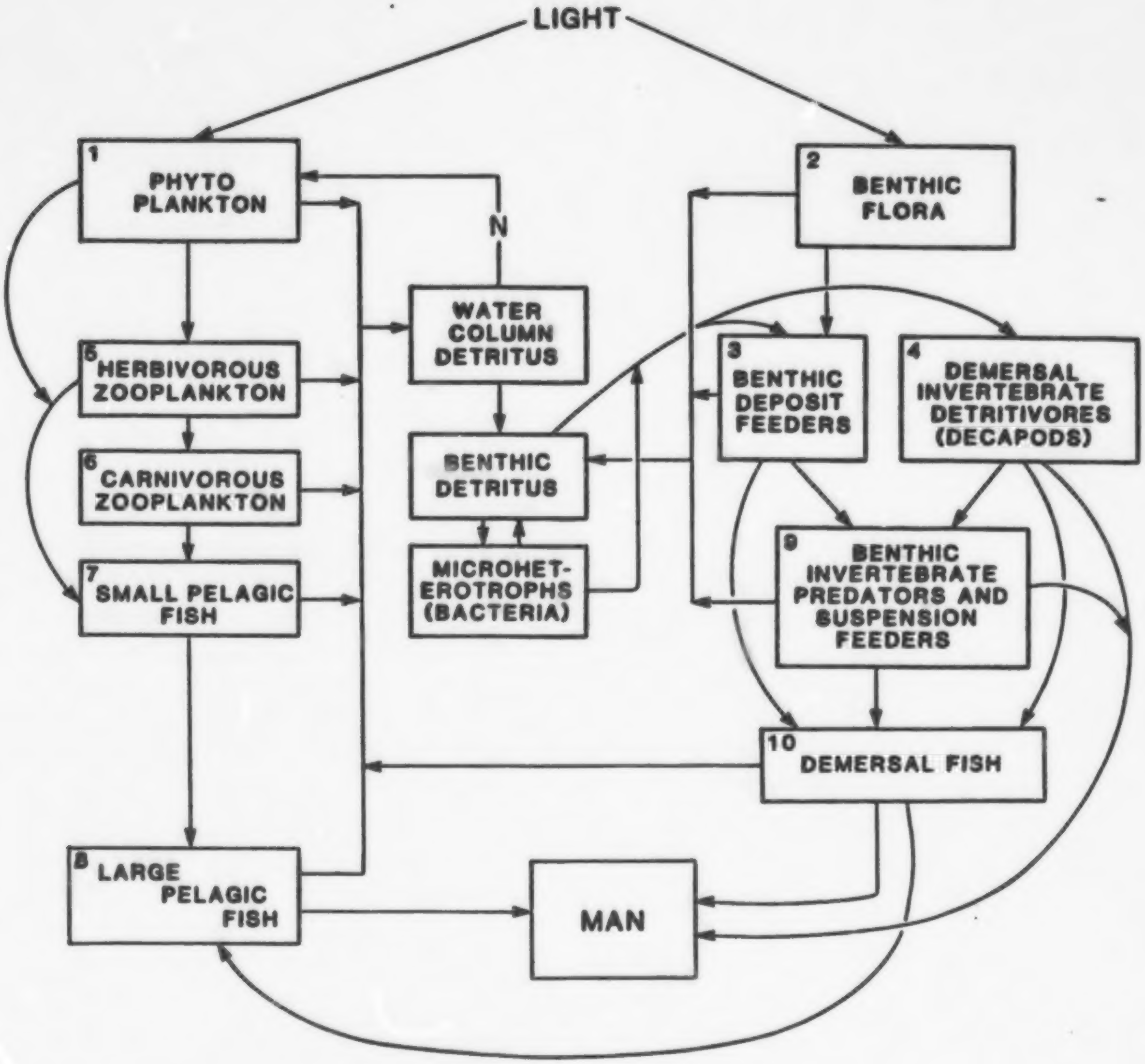
LIGHT
1 PHYTO PLANKTON
2 BENTHIC FLORA
N
WATER COLUMN DETRITUS
3 BENTHIC DEPOSIT FEEDERS
4 DEMERSAL INVERTEBRATE DETRITIVORES (DECAPODS)
5 HERBIVOROUS ZOOPLANKTON
6 CARNIVOROUS ZOOPLANKTON
BENTHIC DETRITUS
7 SMALL PELAGIC FISH
MICROHET-EROTROPHS (BACTERIA)
9 BENTHIC INVERTEBRATE PREDATORS AND SUSPENSION FEEDERS
10 DEMERSAL FISH
8 LARGE PELAGIC FISH
MAN

Figure 3-1. Summary of energy (carbon) flow in ecological model as applied to the Gulf of Mexico.

demersal organisms (#2, 3, 4, 9, 10) only in the bottom layer.

The biomass of compartment i, B_i, is calculated using the following differential equation.

$$\frac{dB_i}{dt} = B_i \; (-FM_i - PM_i - \sum_{j=1}^{n} P_{ij}B_j) \; + \text{GROWTH} \qquad (3\text{-}1)$$

where FM_i is fishing mortality rate, PM_i is pollutant-induced mortality rate, $P_{ji}B_j$ is the predation mortality rate due to predation by compartment j, and GROWTH is growth rate.

For phytoplankton (i=1), growth rate is assumed to be regulated by temperature, light and nitrogen, which are generally thought to be most limiting to phytoplankton in the sea (see Falkowski, 1980). The growth of the benthic flora is regulated by temperature and light, assuming nitrogen to be in plentiful supply. Eppley's (1972) model of algal growth (M_m) as a function of temperature (T) was used for both compartments.

$$M_m = 0.59 \; e^{0.0633 \; T}$$

For light-limited growth, Yoder's (1979) model is combined with Eppley's temperature model:

$$M = M_m \tanh (A_I (I - I_c) / M_m)$$

where M is growth rate (day^{-1}), I is light intensity, I_c is the light intensity where growth rate is zero, A_I is a constant which is a function of temperature, and M_m is as defined above. For nitrogen-limited growth, the Michaelis-Menton formulation is used (as in French, 1984)

$$M = M_m (N / (K_N + N))$$

where N is nitrogen concentration and K_N is a constant which is functionally related to temperature.

Temperature varies sinusoidally over the annual cycle, and cycles typical of each of the three sites, and of the surface and bottom layers, are taken from published data (Franks et al. 1972; Rezak et al. 1983). Incident light intensity varies sinusoidally as well, with a mean of 350 ly/day and amplitude of 150 ly/day (after Rezak et al. 1983). Photosynthetically active radiation is assumed to be 43% of incident light (Yoder, 1979) and light intensity decreases exponentially with depth:

$$I_z = I_s e^{-kz}$$

where I_s is light intensity at the surface, I_z is light intensity at depth = z (m), and k is the extinction coefficient (m^{-1}). The extinction coefficient increases with increasing turbidity and algal concentration in the water:

$$k = k_w + k_c \text{ CHL}$$

where CHL is chlorophyll concentration and k_w and k_c are constants. CHL is assumed equal to 0.01 times phytoplankton carbon concentration ($B_1/100$) and $k_c = 0.03$ (French, 1984). The values of k_w were estimated from Rezak et al (1983) for shelf and offshore sites, with a reasonable value assumed for a turbid estuary such as Mobile Bay (Table 3-2).

Growth of all animal compartments is illustrated in Figure 3-2 and calculated as follows:

$$G_S + G_R = I - R - E - F$$

where G_S is somatic growth rate (day^{-1}), G_R is reproductive tissue growth rate (gonads), I is ingestion rate, R is respiration rate, E is excretion rate (urine) and F is fecal production rate. G_R is assumed a constant fraction of total growth ($G_R = rGs$), excretion a fraction of ingestion ($E = eI$), and fecal production a fraction of ingestion ($F = (1-a) I$, where a = assimilation rate). Thus

$$G_S (1 + r) = I - R - eI - (1 - a) I = (a - e) I - R$$

Table 3-2. Some site characteristics assumed in the model.

Item	Mobile Bay	Shelf	Offshore
Surface layer depth (m)	3	20	60
Bottom layer depth (m)	0	100	1375
Surface temperature:			
mean	23.4	23.4	25.25
amplitude	7.0	7.0	4.25
Bottom temperature:			
mean	23.4	19.5	15.0
amplitude	7.0	3.5	0.0
K_w (m^{-1})	2.23	0.05	0.02
K_n (ug - at/l)	0.50	0.50	0.20

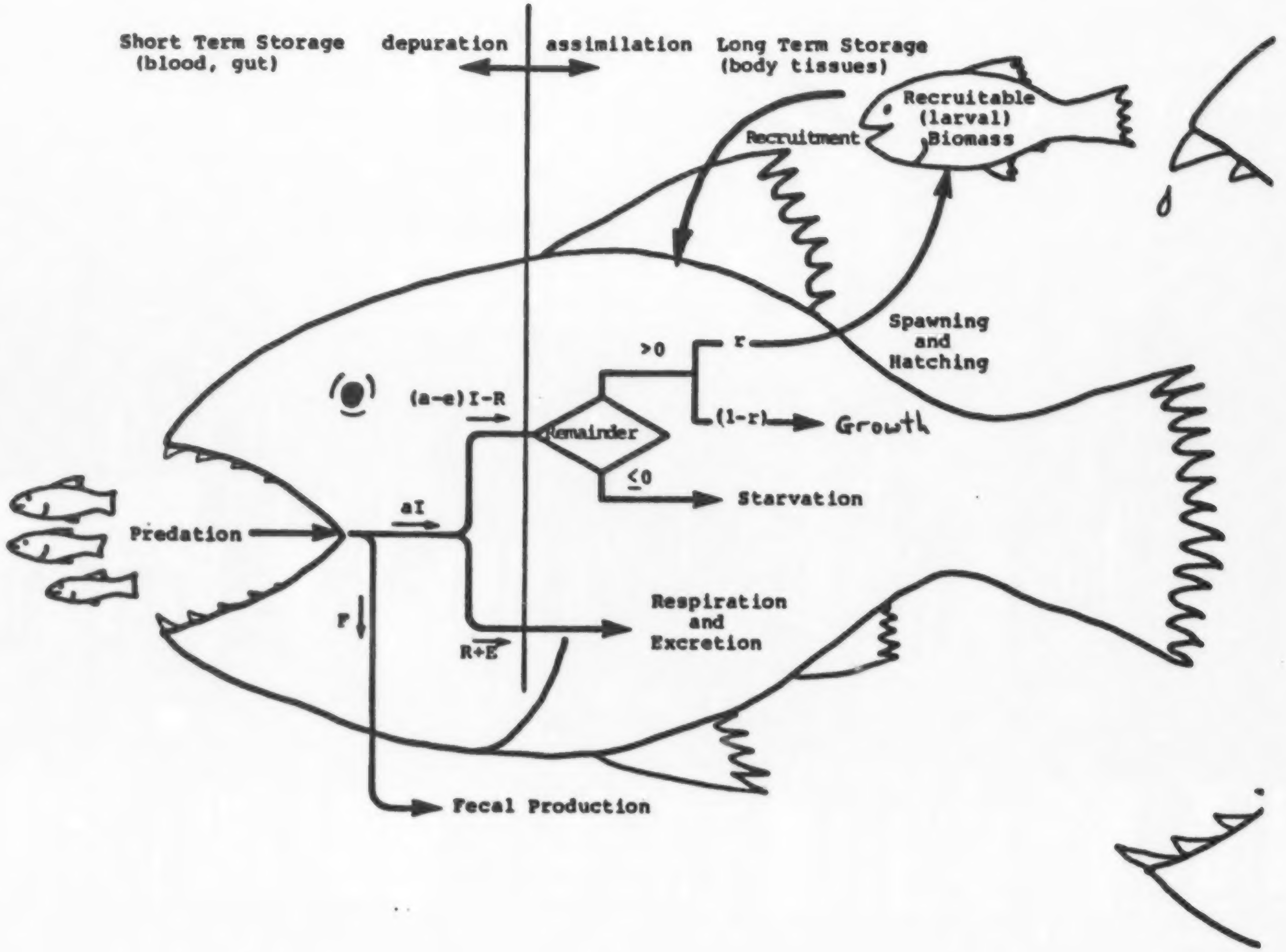

Figure 3-2. Schematic of the physiological submodel for faunal trophic compartments.

Assimilation rates (a) were assumed to be 80-90% for live food and 30% for detritus (Bougis, 1976; Parsons et al. 1977; Browder, 1982; Durbin and Durbin, 1983). Excretion rates (e) were assumed to be 10% (Durbin and Durbin, 1983) and reproduction growth (r) as 1% of somatic growth.

Both ingestion and respiration rates are functions of temperature

$$I = I_o \, e^{Q_I T}$$
$$R = R_o \, e^{Q_R T}$$

where I_o and R_o are rates at zero degrees and Q_I and Q_R are constants. The values of Q_I and Q_R are assumed to be 0.0693 which is equivalent to a $Q_10 = 2.0$, considered average for many biological processes in a variety of organisms (Parsons et al. 1977). Where possible, I_o and R_o were estimated from literature studies of feeding and respiration rates of organisms representative of the specific compartment and model site. Where data were not available, estimates were based on organism size typical of the compartment, since (specific) feeding and respiration rates increase with decreasing organism size (Zenthan, 1953; Bougis, 1976; Jones and Johnston, 1977). Respiration rates of zooplankton were obtained from studies summarized by Bougis (1976), of menhaden (compartment #7, Table 3-1) from Durbin and Durbin (1983), of shrimp (#4) from Bishop et al. (1980), of tuna (#8, Table 3-1) from Brill (1979), of pinfish (#10) from Hoss (1974).

Respiration rates for compartments 3 and 9 are based on assumed organism size.

Available data on ingestion rates is much more limited. Reliable estimates are only available for zooplankton (#5 and #6) from Deason (1980) and work summarized by Bougis (1976) and Parsons et al. (1977) and for menhaden (#7) from Durbin and Durbin (1983). For other compartments, rates are based on size and fit to available standing stock data for the site in question. For all (animal) compartments, ingestion may not exceed a maximum ration ranging from 190% of body weight per day for herbivorous zooplankton (Deason, 1980) to 20% per day for fishes.

All fecal material flows to the water column detritus pool, where it is either remineralized yielding inorganic nitrogen (feeding back to the phytoplankton compartment), or it settles to the sediments. Benthic detrital carbon is respired by bacteria and ingested by benthic organisms (compartments 3 and 4).

All animal compartments produce eggs according to an annual spawning cycle typical of the Gulf of Mexico. Eggs are assumed to hatch instantaneously. The larvae of compartments 3, 4, 5, 6 and 9 (invertebrates) function as herbivorous zooplankton (#5) until recruited to the adult population. Larvae of the three fish compartments (icthyoplankton) are considered carnivorous zooplankton (#6) until recruited. Recruitment rate increases exponentially with temperature ($Q_{1}0 = 2.0$) and is assumed to occur after 100 days for invertebrate larvae and 1000 days for fish at zero degees C.

Predicted biomass levels for the three sites are illustrated in Figures 3-3 through 3-5. Units are mg C/m^2, where 1 g C is approximately equivalent to 2.5 g dry weight or 17.5 g wet weight (Parsons et al. 1977, Bougis 1976). Standing stock estimates are most readily available for comparison for the shelf region from literature studies (Table 3-3). Browder (1982) estimates phytoplankton biomass to be about 45 mg C/m^3. The shelf simulation reported here reflects an annual average of 48 mg C/m^3. Primary productivity in the simulation is 134 g C m^{-2} $year^{-1}$ which is a reasonable estimate for a shelf region (Smayda, 1976) but lower than Sklar's (1976) estimate for the near-shore Louisiana shelf (Table 3-3). Biomass levels in the Mobile Bay (Table 3-4) and offshore simulations (Table 3-5) are similar to shelf levels but primary production per unit volume decreases with increasing euphotic zone depth. On an areal basis offshore production is 133 g C m^{-2} $year^{-1}$, which is also reasonable (Smayda, 1976, Table 3-5).

Mobile Bay simulation predicts an annual productivity of 36 g C/m^2, the relatively low value owing to the shallow water column (3 m). No field estimates of annual primary productivity levels have been found in the literature for this bay but others shallow bays have primary productivities near this value (Table 3-4).

Phytoplankton in the shelf and offshore sites are limited by nitrogen year round, whereas phytolankton in the Mobile Bay simulation are at times both nitrogen and light limited. This difference is due to the higher nigtrogen supply rates in the

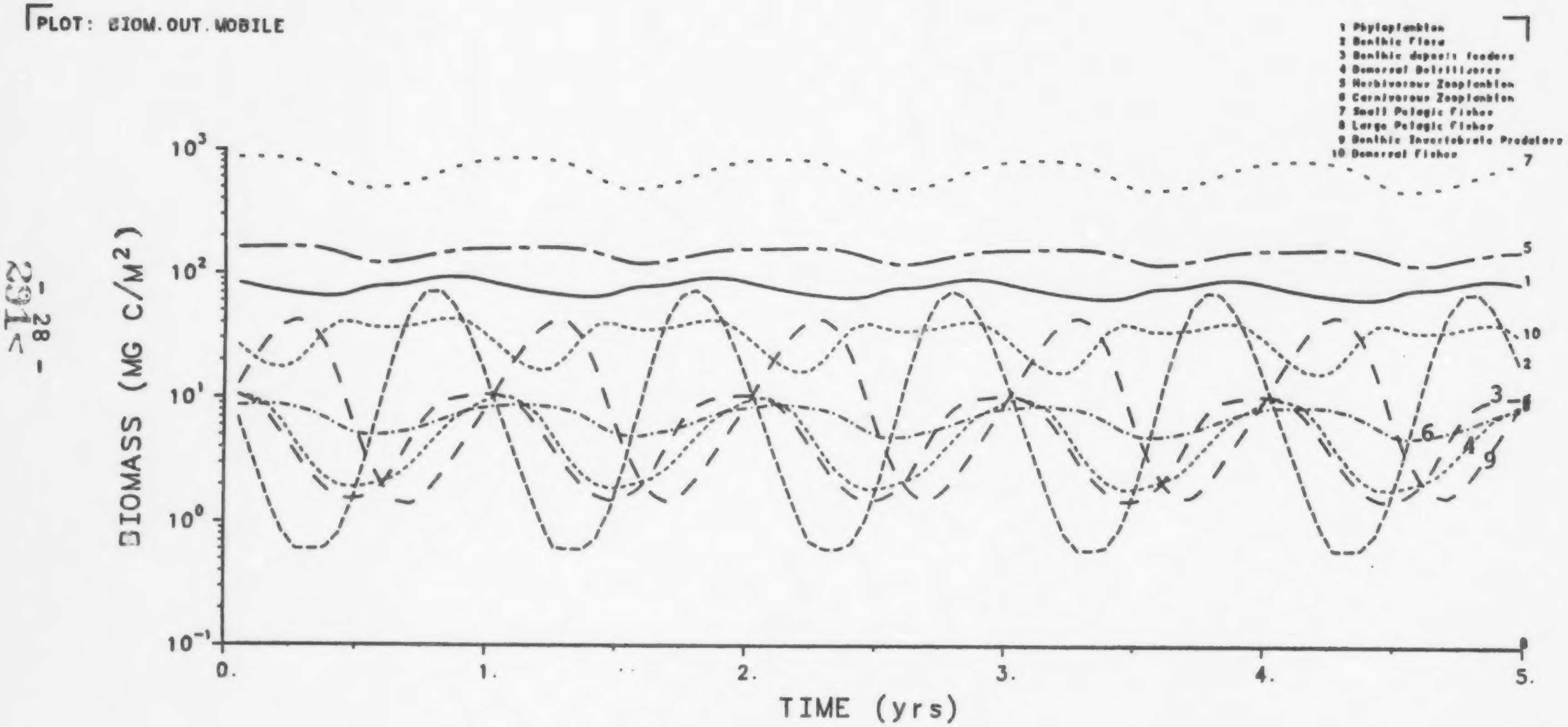

Figure 3-3. Biomass levels in the Mobile Bay ecosystem at background (non-inhibitory) levels of pollutants.

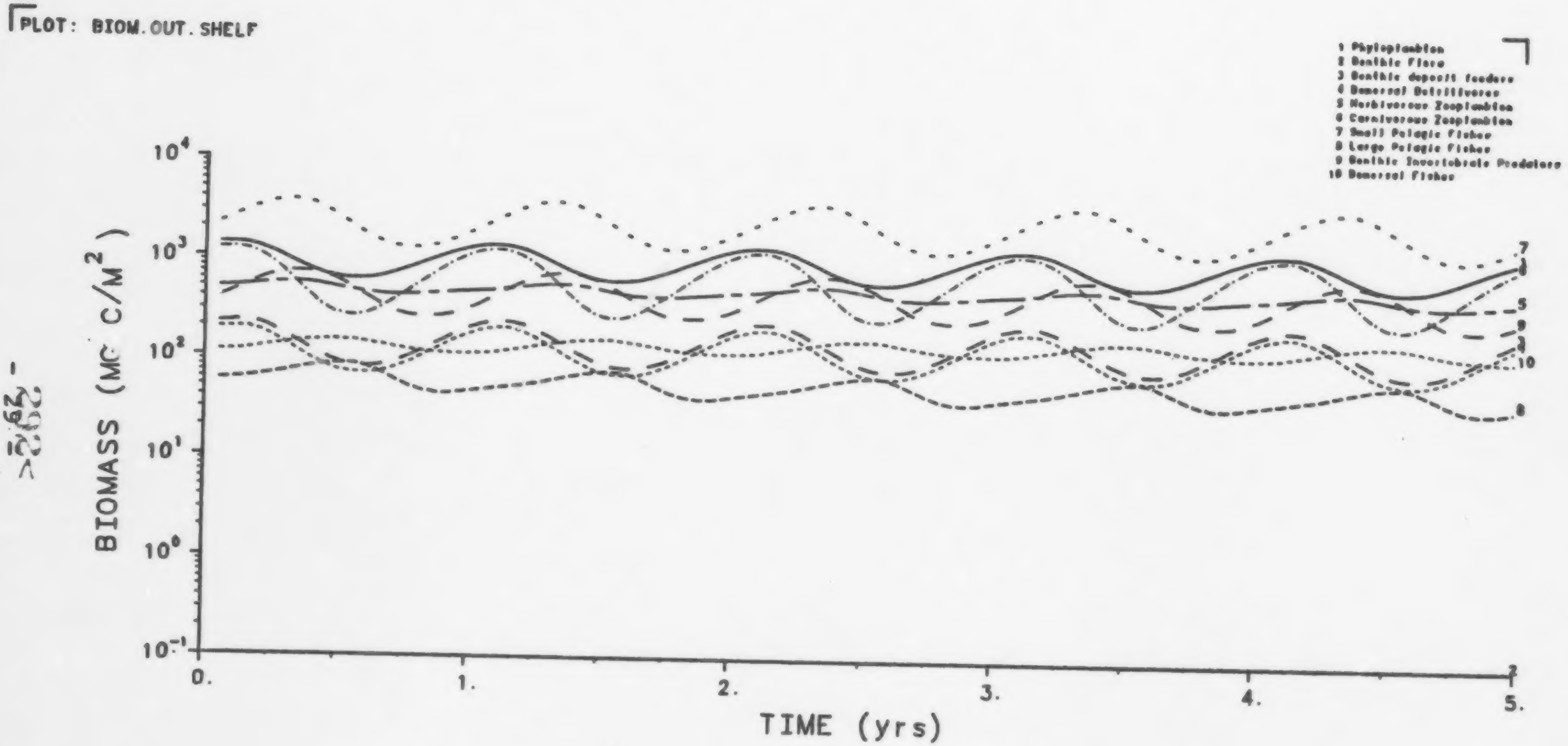

Figure 3-4. Biomass levels in the shelf ecosystem at background (non-inhibitory) levels of pollutants.

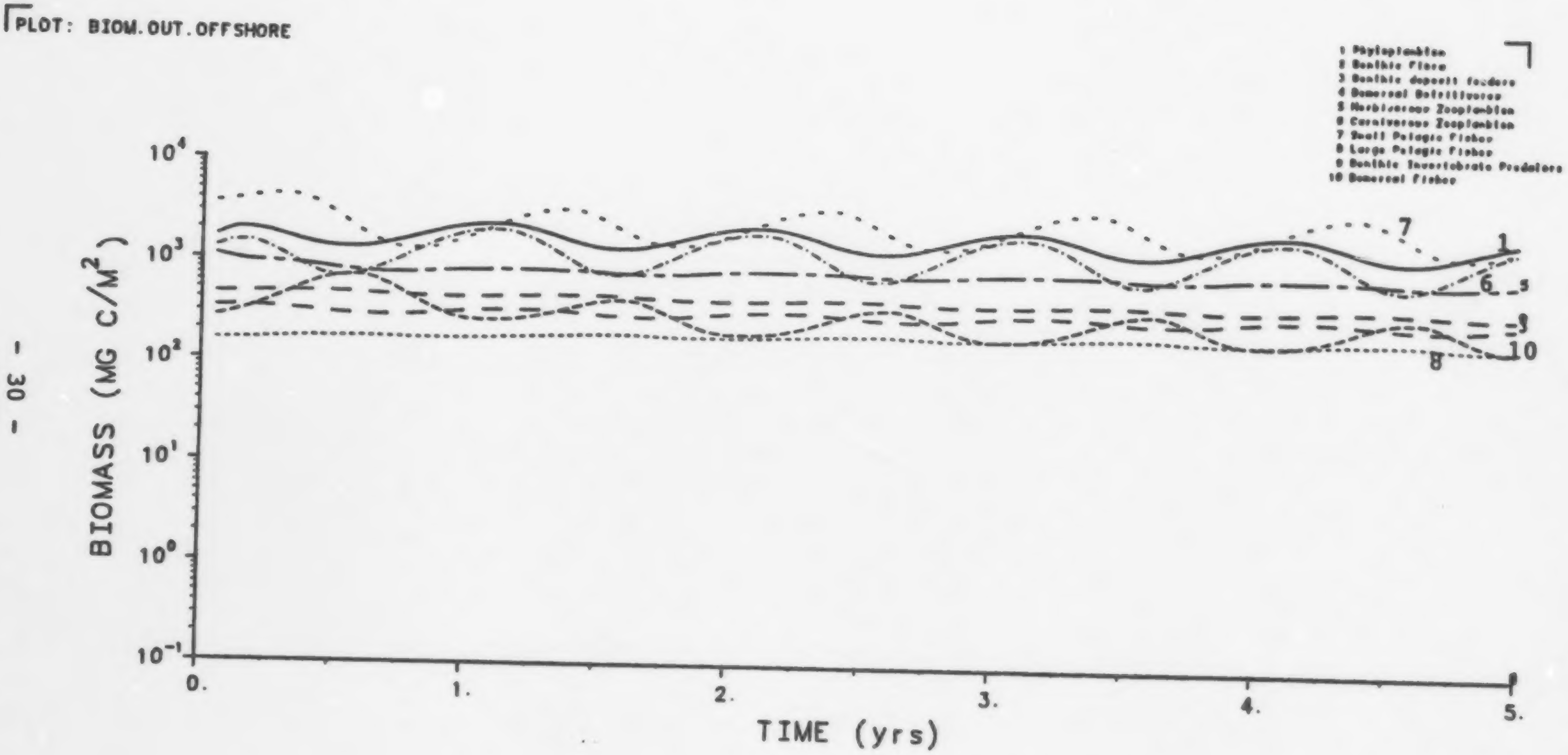

Figure 3-5. Biomass levels in the offshore burn zone ecosystem at background (non-inhibitory) levels of pollutants.

Table 3-3. Comparison of model estimates of continental shelf standing stock (mg c/m^2) and primary productivity (g C/m^2.yr) with observations for Gulf of Mexico or similar shelf regions.

Biota	#	Model	Observed	Reference
Phytoplankton	1	960	900	Browder, 1982
Annual primary production		134	100-350 266	Smayda, 1976 Sklar, 1976
Zooplankton	5+6	1180	120	Browder, 1982
Pelagic fish	7+8	2456	900 1200	Rezak et al, 198 Browder, 1982
Benthos	3+4+9	634	3200	Browder, 1982
Decapods	4	126	32 (shrimp only)	Browder, 1982
Demersal Fish	10	138	804	Browder, 1982

Table 3-4. Comparison of Mobile Bay model estimates of standing stock (mg C/m^2) and primary productivity (g C/m^2.yr) with observations of other shallow estuaries in the Gulf region.

Biota	#	Model	Observed	Reference
Phytoplankton	1	79		
Annual primary production		36	50	Smayda, 1976
Zooplankton	5+6	159		
Pelagic Fish	7	600	520	Constanza et al, 1983
Benthos	3+4+9	25.8	3800	Constanza et al, 1983
Decapods	4	5.1	4.7 (shrimp)	Loesch, 1976
Demersal Fish	10	32		

Table 3-5. Comparison of offshore model estimates of standing stock (mg C/m^2) and primary productivity (g $C/m^2.yr$) with observations for similar open ocean areas.

Biota	#	Model	Observed	Reference
Phytoplankton	1	1800	1200	French et al, 1983
Annual primary production		133	100	Smayda, 1976
Zooplankton	5+6	2100	900	Rezak et al, 1983
Pelagic fish	7+8	2694		
Benthos	3+9	720		
Demersal fish	10	162		

shallow system as well as to higher turbidity. Phytoplankton in the mixed layer of the deep ocean are generally considered to be nitrogen limited (e.g., Dugdale, 1967).

Estimates of zooplankton biomass range from about 6 mg C/m^2 (Browder, 1972) to 45 mg C/m^3 (Rezak et al. 1983) for the Gulf coast region. The model simulations predict levels at the upper end of this range (53, 59 and 35 mg C/m^3 for Mobile Bay, shelf and offshore, respectively). Browder's (1982) estimates of pelagic fish biomass from landings are on the order of 1200 mg C/m^2. The shelf and offshore simulations predict 2456 mg C/m^2 and 2694 mg C/m^2 respectively, or about a factor of two higher. In Mobile Bay, the model predicts pelagic fish density to be 600 mg C/m^2, which compares well with an estimate of 520 mg C/m^2 for Mississippi delta estuaries by Costanza et al. (1983).

Benthic biomass levels (sum of #3, 4 and 9) of 634 mg C/m^2 and 720 mg C/m^2 in the shelf and offshore simulations respectively are much lower than Browder's (1982) estimate of about 3200 mg C/m^2 for a shelf average over 0-93 m depth. The greater depth of the two simulations (100 m and 1375 m) may account for this difference, as benthic biomass is known to decline sharply with water depth (Parsons et al. 1977). Benthic (and demersal fish) standing stocks are also very low in the Mobile Bay simulation (44 mg C/m^2), but higher levels could not be supported by simulated primary production. Inputs of organic matter from rivers and marshes are not explicitly simulated in this model and their omission may be the cause of reduced benthic biomass levels in the Mobile Bay simulation. This consideration is important in Mobile Bay and on the shelf in the Mississippi River plume, not only in terms of productivity estimates, but also for estimates of suspended particulate matter concentration, and pollutant adsorption/dissolution dynamics.

Seasonal cycles are evident in plots of biomass for all three sites (Figures 3-3 to 3-5). For most compartments these are driven by the seasonal cycling of the major prey or predators and of spawning and therefore recruitment. For example, in the case of Mobile Bay, the strong seasonal cycle of the benthic flora is due primarily to the seasonal change in grazing pressure from the benthos (compartment #3) and the biomass plots show the typical predator-prey relationship of a predator's peak following the peak in prey biomass. Compartment #9, which preys mainly on #3 and #4, peaks after their prey in both Mobile Bay and shelf simulations. Offshore, where water depth is 1375 m, seasonal cycles are very weak in the benthos, as would be expected.

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4. Pollutant Exposure - Biological Response Relationships

4.1 Methods and Data Sources

Two types of impact can be estimated by the model system:

(1) bioaccumulation,
(2) biomass reductions through lethal or sublethal toxic effects on parent stock or recruitable biomass.

The two may be coupled, if bioaccumulation results in pollutant levels which are toxic in higher trophic compartments. Sublethal chronic effects, through changes in reproductive rates for example, can be simulated with this model system. Such effects have not been included here because the magnitudes remain highly speculative.

Bioaccumulation is computed in parallel with the physiological model discussed in Section 3, which shows metabolic processes associated with both short term and long term storages. Figure 4-1 gives a schematic for the bioaccumulation calculations within a given compartment. The pollutant concentration in short term (blood) storage for the i^{th} compartment is assumed as a minimum to be in chemical equilibrium with the dissolved phase in the water (or sediments for benthos). If the pollutant concentration in the food exceeds that in the environment the short term storage is set equal to this new level. This is because equilibrium processes across gill tissues are rapid relative to the one day timestep on which the model typically runs. Long term storage (fats and proteins) is connected to short term storage as shown in Figure 4-1, by an assimilation rate f_1 and a depuration rate f_2. Elevated concentrations in the blood (short term storage) relative to the

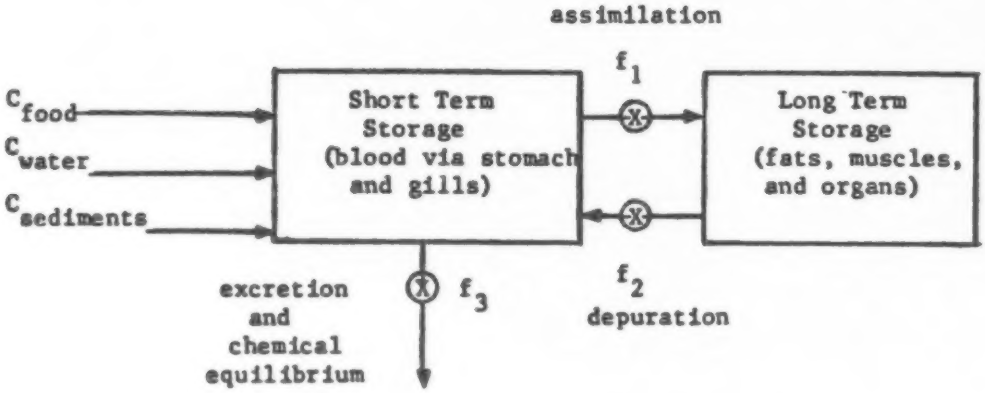
assimilation
f_1
C_{food}
C_{water}
$C_{sediments}$
Short Term Storage (blood via stomach and gills)
Long Term Storage (fats, muscles, and organs)
excretion and chemical equilibrium
f_3
f_2
depuration

Figure 4-1. Metabolic submodel schematic for bioaccumulation estimation.

environment result in pollutant loss to the environment at the rate f_3. The rates f_1, f_2, and f_3 are both pollutant and species (trophic compartment) specific.

PCBs enter animals primarily by uptake directly from the water and by absorption from food through the intestinal wall. Weininger (1978) and Thomann and Connolly (1984) have shown the latter to be the more important accumulation mechanism. Accordingly, accumulation of PCBs occurs mainly from food in the model. For each (animal) compartment, short-term storage (= blood or gut) is assumed at minumum to be in equilibrium with the concentration of PCBs in the water (or sediments for benthic organisms). If the food contains a higher concentration of PCBs, short term storage is equivalent to this value. Organisms take up PCBs at a compartment-specific assimilation rate into long-term storage (tissue, particularly fats) and excrete PCBs back to the blood (and eventually the water) at a compartment-specific depuration rate.

Assimilation rates of PCBs from food are not well known. From a study by Wyman and O'Connors (1980), Thomann and Connolly (1984) calculated an assimilation rate of 20% for Acartia tonsa fed PCB-contaminated phytoplankton. This rate is assumed for zooplankton compartments #5 and #6 as well as for benthic compartments #3 and #4. Thomann and Connolly (1984) indicate that assimilation rates for fish are higher, perhaps 70%. Smith et al. (1980) measured assimilation rates of 61 trout. A range of rates were tried in the model simulations. PCB depuration rates are more readily available for a variety of organisms and were estimated from data compiled by Thomann (1978) and Thomann and Connolly (1984), and several values tested for each compartment. A depuration rate of 0.01/day was calculated from experiments on spot by Hansen et al. (1971). This value is assumed for demersal (#10) and small pelagic (#7) fish.

Marine toxicity data for EDC is relatively meager. Information in the literature (Veith et al. 1980, EPA 1980) suggests that this pollutant does not bioconcentrate (bioconcentration factor of 2, versus 10,000 for Aurochlor 1254), that it is not acutely toxic to most marine organisms. It is also relatively volatile. Veith et al. (1980) measured depuration rates of EDC in sunfish at 0.25-0.50 per day. Values in this range were used in the model for fish. Invertebrates were assumed to excrete EDC at higher rates than do fish, as is the case for PCBs. Assimilation rates are not available for EDC, but should be about 40-50% per day if bioconcentration factors in fish are on the order of 2.0, as was measured by Veith et al.

Biomass reductions through pollutant-induced mortality are simulated through the second term ($PMRT_i$) in Eq. 3-1. Bioassay results are used to derive mortality response curves for each pollutant. In general, the time and space scales of the problems addressed here inherently focus on long term chronic rather than short term acute toxic effects. Our understanding of the full range of sublethal disruptive responses at each trophic level is poor relative to our knowledge of acute toxic effects. It is therefore necessary in general to extrapolate low level effects from acute bioassay data, and include what little chronic bioassay data may be available.

The approach taken here to modeling pollutant-induced mortality is that organisms subjected to a given pollutant loading show little response up to some limiting value. After this threshold value is reached, some fraction of the population incurs physiological injuries and dies. A survivorship ratio, SR, can be defined as the number of survivors, N_t, in some population of which was of size N_o before pollutant exposure:

$$SR = N_t/N_o \tag{4-1}$$

where N_o equals the original population at time $t = 0$. The mortality ratio, MR, is then simply:

$$MR = 1 - SR \tag{4-2}$$

The population mortality ratio as a function of pollutant concentration can also be expressed as

$$MR = \exp\,(K * LC - C_o) - 1 \tag{4-3}$$

The relevant parameters include K, a trophic level and pollutant-specific mortality coefficient, LC, the lethal concentration for some known mortality ratio (e.g. LC_{50}), and C_o, the threshold concentration.

4.2 Results

A 96-hour LC_{50} concentration is the traditional benchmark in toxicology studies. The LC_{50} is defined as that concentration required to induce 50% mortality in the original population over a 96-hour period. To determine the appropriate LC_{50} values for the pollutants of interest, a literature review focused on the EPA Ambient Water Quality Criteria documents (U.S. E.P.A., 1980) for the respective pollutants. Background pollutant mass concentrations were compiled from several sources and chosen to represent C_o values, below which no mortality is induced. Solution of Equation 4-3 for the trophic-pollutant specific mortality coefficent with substitution of the LC_{50} and C_o values allows the generation of the desired exposure-response curves (Figures 4-2 and 4-3).

It should be noted that the treatment of background concentrations as threshold values leads to a possible overestimation of chronic toxic effects on populations, assuming that present background levels have no adverse effects. overestimation is probably desireable given the uncertainty associated with biological effects modeling. Although not yet complete, work is now underway to begin to quantify this uncertainty. The methodology used here also does not account for sensitivity variability within trophic compartments, but is intended to reflect an exposure-mortality response which is average or typical for each group of biota.

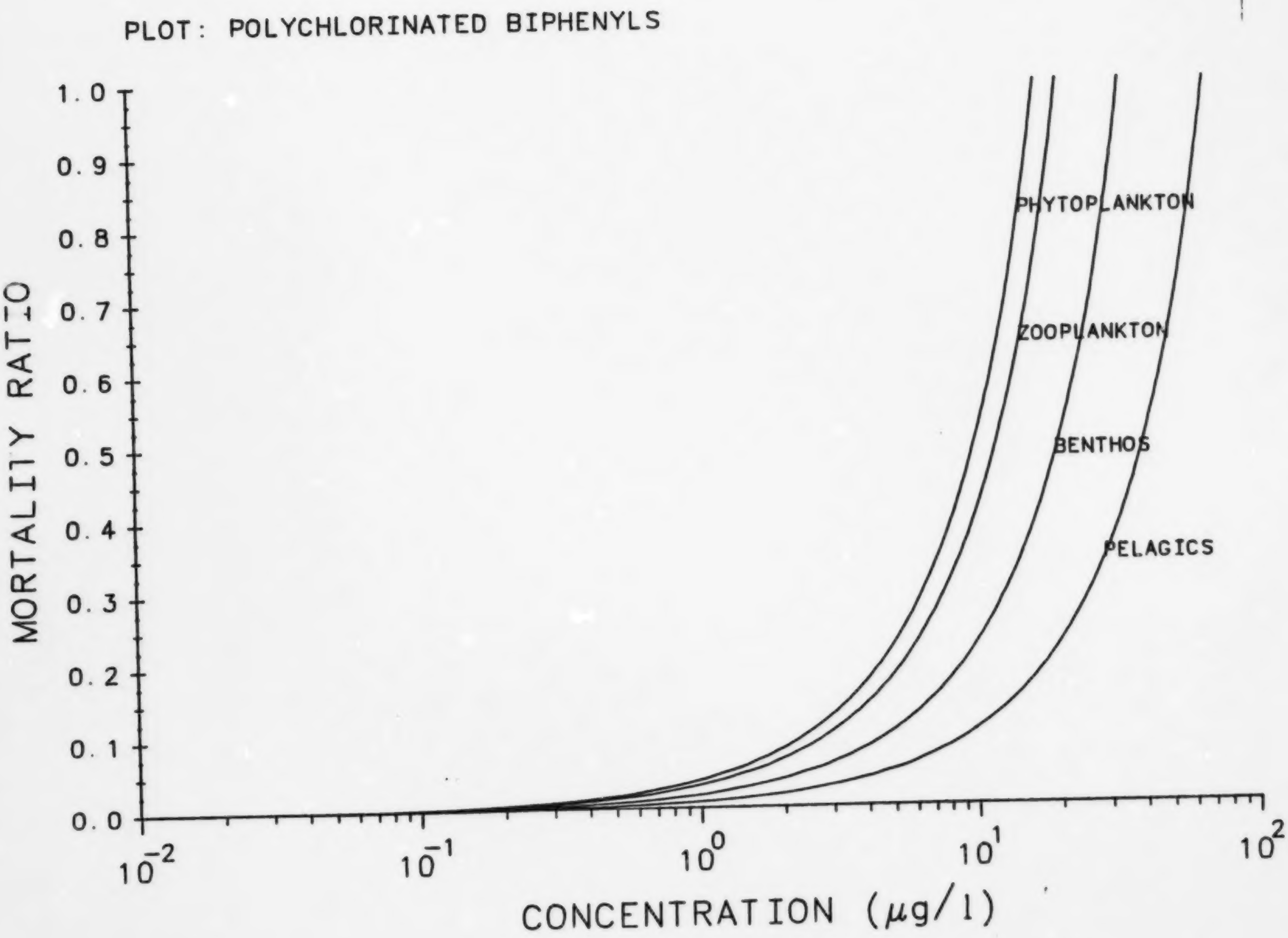

Figure 4-2. Mortality as a function of ambient PCB concentration (ppb). Mortality ratio is the portion dead after four days at the given concentration.

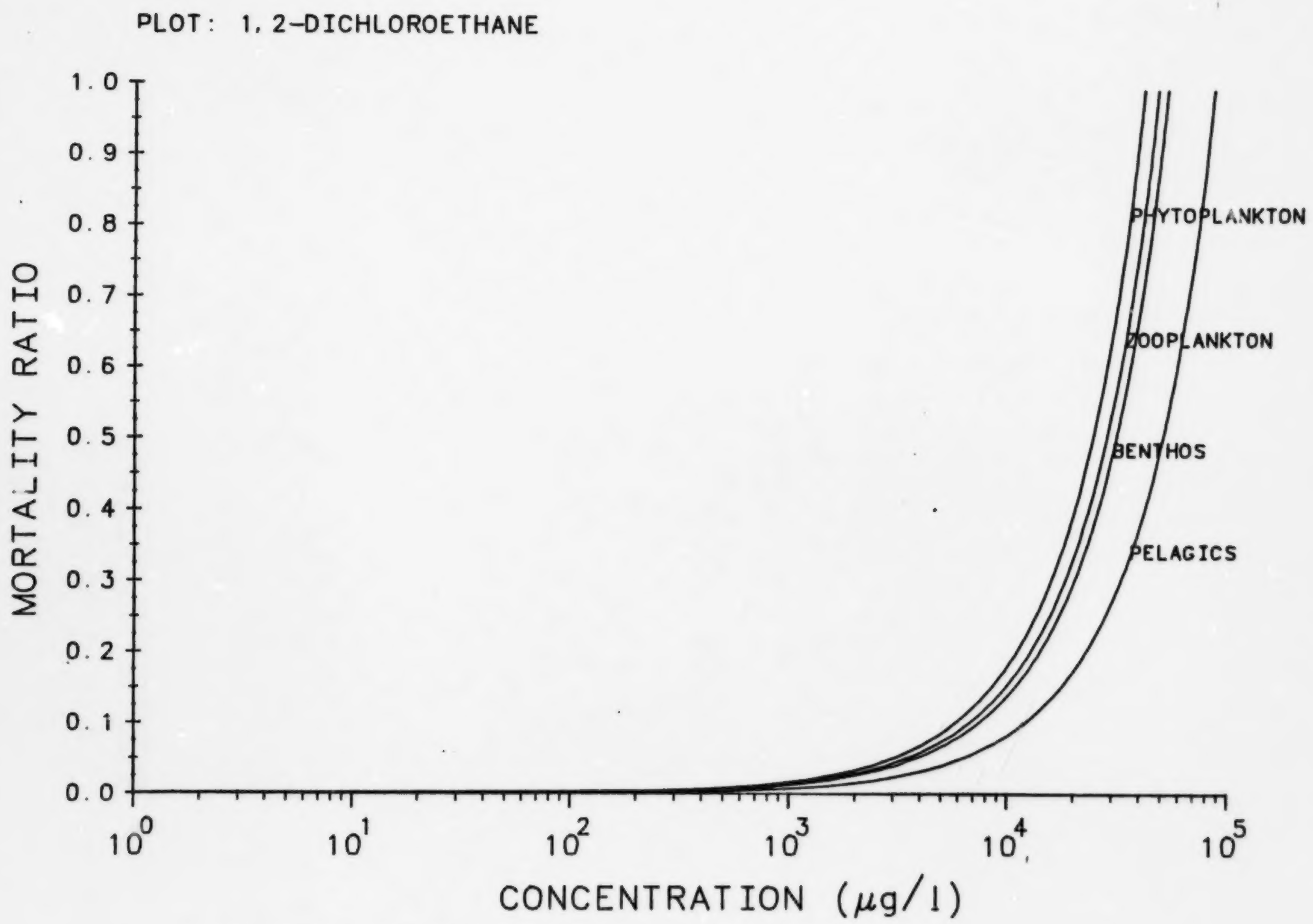

Figure 4-3. Mortality is a function of ambient EDC concentration (ppb). Mortality ratio is the portion dead after four days at the given concentration.

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5. Marine Ecosystem Effects Due to Stack Emissions

5.1 Overview

The entry of atmospheric pollutants into the oceans is mediated by the surface layer. The extreme surface, or microlayer, is of special interest because very high levels of biological activity and pollutant concentrations have been observed in it. Possible relationships between the microlayer and the rest of the marine ecosystem, and potential effects of pollutants in the microlayer, are discussed in the following section. The time frame of the present study has not permitted the incorporation of the microlayer into the ecosystems model, so the discussion is necessarily qualitative. Conclusions are further limited by the fact that microlayer processes are incompletely understood.

The modeling analysis discussed below is based on an incinerator stack emission rate of 1 kg/hr, consistent with the rate used in the atmospheric modeling work, Appendix G. This corresponds to a burning efficiency of about 0.9999, whereas actual efficiency of 0.999999 is expected. The emission rate, and impact areas and concentrations (see summary Figure 5-5) are therefore overestimated by a factor of about 70. Although the model system predicts measurable localized changes in body burdens at the high emission rate modeled, Figure 5-5 shows that the 1 ppb level will not be exceeded within 50 years of continuous incineration at the lower expected efficiency of 0.999999, and ecosystem effects are therefore expected to be negligible.

5.2 Description of the Microlayer

The sea surface microlayer is usually operationally defined as the first 100 um-thick layer of the sea at the air-water interface. The definition is a consequence of commonly-used sampling methods. Most often, sampling is accomplished by means of a screen (Garrett 1965), a perforated telflon plate (Larsson et al. 1974), or a rotating drum (Harvey 1966), to which the surface film adheres. These and other sampling methods are reviewed and compared by Hatches and Parker (1974), Liss (1975), Daumas et al. (1976), Hamilton and Clifton (1979), Van Vleet and Williams (1980). Several of these authors have emphasized that the microlayer is actually much thinner than can be successfully sampled. Theoretically, the microlayer could be considered to be a mono-molecular film or a layer where water molecules vary from bulk liquid orientation. The actual characteristics of the microlayer depend on the depth of the surface layer considered.

A number of investigations have shown the microlayer to be relatively enriched in dissolved and particulate organic matter (Garrett 1965, Harvey 1966, Williams 1967, Duce et al. 1972, Liss 1975) including pollutants such as T residues (Seba and Corcoran 1969), PCBs and hydrocarbons (Duce et al. 1972), nutrients (Goering and Menzel, 1965, Williams 1967, Taguchi and Nakajima 1971, Liss 1975), metals (Piotrowicz et al. 1972), and organisms including bacteria, protozoans, phytoplankton and zooplankton (Taguchi and Nakajima 1971, Marumo et al. 1971, Liss

1975, Sieburth et al. 1976). The sea-surface microlayer has been shown to be enriched by factors of from 2 up to 6 orders of magnitude relative to the bulk liquid, depending on the sampling method used and the location and time of sampling.

Liss (1975) summarizes the mechanisms of transport into and out of the microlayer as follows. Material enters the microlayer from the air via rain or dry deposition (mainly on particles such as dust) and from spills (anthropogenic). Material returns to the air in the form of wind-driven aerosol, through evaporation and by ejection from bursting bubbles. Microlayer material is dispersed into sub-surface water by dissolution, wave action and sinking. Accumulation in the surface layer from subsurface sources occurs via advection, diffusion and (most importantly) via bubbles.

The most important enrichment mechanisms include the following (see reviews by Liss 1975, Duce and Hoffman 1976). Surface-active molecules which contain hydrophobic and hydrophilic portions collect at the air/sea interface. These materials are largely lipids, fatty acids and hydrocarbons. These surfactants are also scavanged by bubbles, and largely left at the surface when the bubbles burst. Ionic materials, including nutrients and metals, adsorb on these surfactants and are thus concentrated in the microlayer as well. Organisms may be concentrated in the surface layer due to the concentration of food. Bacteria feed on the organic matter, and their predators follow (Sieburth et al. 1976). Organisms may also be physically concentrated there by bubble-scavanging (Nishizawa, 1971).

Pollutants such as DDT residues and PCBs, accumulate in the microlayer owing to their lipophilic nature and their low solubility in water.

The organisms in the surface microlayer are not only in greater abundance, but form statistically different communities from the subsurface layer (Taguchi and Nakajima 1971, Sieburth et al. 1976). Taguchi and Nakajima have observed certain phytoplankton and zooplankton species found only in or preferentially in the surface microlayer. The phytoplankton of the microlayer were photosynthesizing, although inhibited by the high irradiance and ultraviolet light levels at the surface. Zooplankton species were more abundant at night (Taguchi and Nakajima, 1971). Therefore, the surface microlayer represents a distinct community and a concentration of biological activity and food for larger organisms.

5.3 Atmospheric Pollutant Effects

The role of the microlayer community in concentrating and dispersing pollutants is poorly known. Bair (1972) measured bacterial activity on hydrocarbon films and compared it to physical dispersion mechanisms. He concluded that bacterial breakdown was slower than either bubble-breaking ejection into the atmosphere or ultraviolet breakdown as dispersal mechanisms. Thus physical mechanisms, primarily wave breaking, probably represent the major mechanisms for entrainment of microlayer pollutants into the underlying water column.

PCB and EDC

The fact that organic and heavy metal pollutants concentrate in the microlayer suggests that atmospheric pollutant effects will be highest on the microlayer community. Ofstad and Lunde (1978) report surface film concentrations of PCB up to 10^6 times higher than concentrations in the underlying water. The intensity of biological activity will result in faster accumulation into the food chain via the microlayer community than via the bulk pelagic community. Seba and Corcoran (1969) have argued that pollutants enter the food chain mainly through microlayer planktonic organisms. The magnitude of this effect is poorly known and remains unquantified, but a reasonable hypothesis is that ichthyoplankton and small pelagic fish rely on the highly productive microlayer for some fraction of their total dietary intake. Depending on the size of this fraction, significant bioaccumulation could occur via this pathway, a hypothesis which could be quantitatively investigated through modification of the ecosystem model.

The atmospheric transport model described in Appendix G has been coupled to the ocean transport model to quantify the distribution in the ocean of PCBs from the incinerator stack. Distribution and effects of EDC have been neglected here since even very large spills of this chemical are shown in Section 6 to have negligable effects due to high solubility and volatility. (Note that the PCB emission rate for the atmospheric model was set at 1 kg/hr. Since actual efficiency of 0.999999 is expected, the concentration levels discussed here are high by a factor of approximately 200.)

Table 5-1 lists values of parameters necessary as input to the atmospheric model.

Figure 5-1 shows the spatial distribution averaged over the upper water column of PCB deposited to the sea surface and settling according to an observed settling velocity distribution for natural aggregrates in sea water. During incineration, water column concentrations reach, at most, 0.1 parts per trillion in the surface water and decline quickly once the 11 day burn is complete. These concentrations are two orders of magnitude below background concentrations of PCB, and so would have no measurable effect on the ecosystem. Figure 5-2 shows the distribution on the sea floor of PCBs deposited on the sea surface by the incinerator plume. This figure is based on the assumptions that PCB settling velocities in the water column are governed by ambient particle settling velocities through adsorption, and that the microlayer residence time is negligible. Figure 5-3 shows, for purposes of comparison, the effect of a 7 day microlayer residence time on ultimate seafloor distribution. Within the range of 0 - 7 days, residence time in the microlayer has only a small effect on the ultimate distribution of PCBs in the sediments.

Figure 5-4 shows the seafloor area which would be contaminated by greater than 1 ppb PCBs after 5, 10, 20 and 50 years. (Note that the PCB emission rate for the atmospheric model was set at 1 kg/hr. Since actual efficiency of 0.999999 is

Table 5-1. Parameter values for atmospheric model applications.

Parameter Name	Symbol	Value
von Karman constant	k	0.4
stability coefficient	c	7.8
Coriolis parameter	f	7.3×10^{-5}
particle diameters	D_i	(micrometers)
-arsenic		1
-cadmium		2
-chromium		3
-nickel		1.5
-generic metal		1.9
-PCB		10
particle densities	S_i	(gm/cm^3)
-arsenic		5.7
-cadmium		8.7
-chromium		7.1
-nickel		8.9
-generic metal		7.6
-PCB		1.5
molecular weights	M	(gm/mole)
-arsenic		75
-cadmium		112
-chromium		52
-nickel		59
-generic metal		75
-PCB		326
atmospheric boundary layer		
-summer		1.5 km
-winter		0.75 km
diffusivities	Kx,Ky	1000 m^2/sec
gaseous PCB deposition rate		0.1 cm/sec
gaseous/particulate PCB mass emission ratio		9

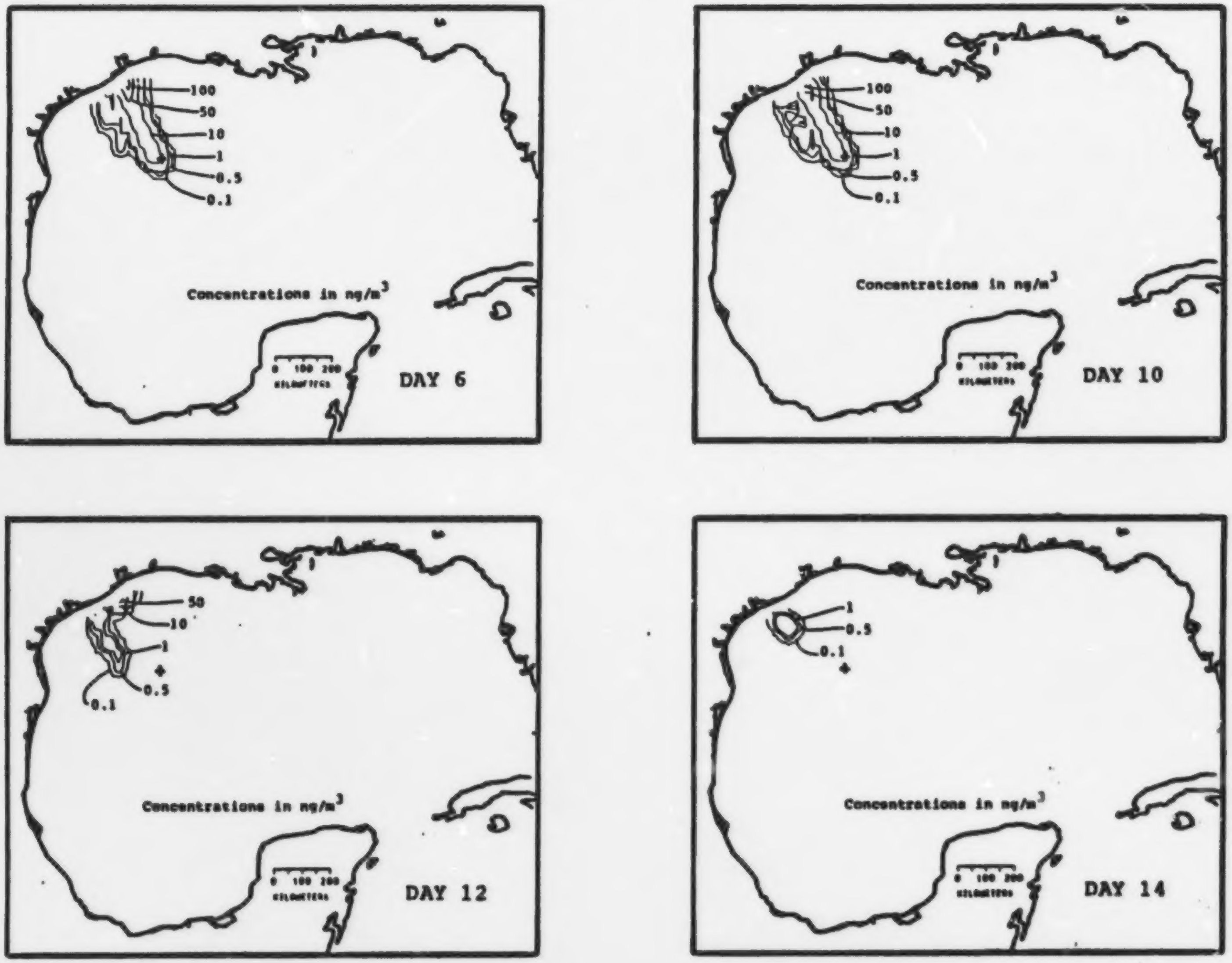

Figure 5-1 Time series of PCB concentrations in upper (60 m) water column, including microlayer, due to an incinerator source strength of 1 kg/hr for an August scenario. a) Day 6 after start of burn, b) Day 10, c) Day 12, d) Day 14

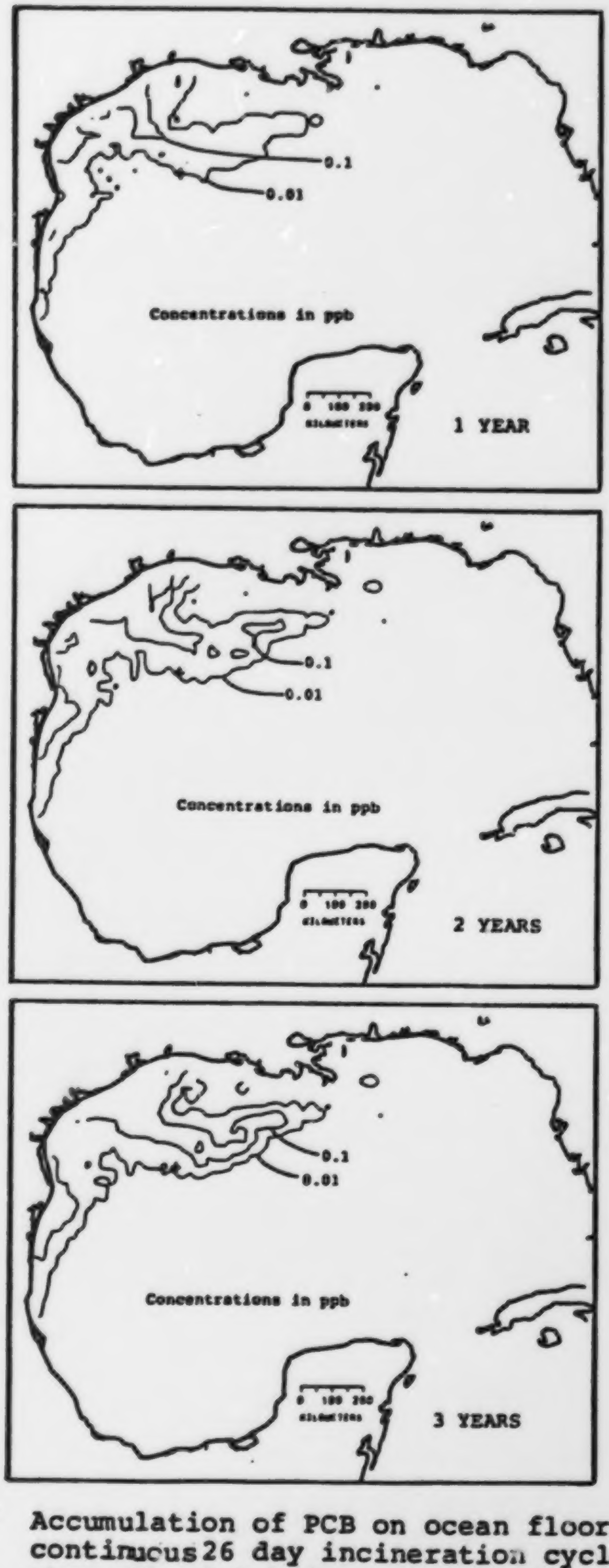

Figure 5-2 Accumulation of PCB on ocean floor due to continuous 26 day incineration cycle, source strength of 1 kg/hr, microlayer residence time of 0.0 seconds. a) End of 1 year, b) end of 2 years, c) end of 3 years.

expected, the concentration levels discussed here are high by a factor of approximately 200.) Burning is assumed to begin every 26 days and is continuous year round. The area of seafloor containing more than 1 ppb in the surface sediments increases rapidly in the first few years, leveling off after 50 years of burning at about 200,000 km^2 (Figure 5-5). Figure 5-5 shows average concentrations of PCBs in the sediment within the 1 ppb contour as a function of time.

The contaminated area of seafloor is partly on the Texas shelf and partly offshore (Figure 5-4), and so both the shelf and offshore ecosystem models were run with sediment concentrations simulated as in Figure 5-5. Water PCB concentrations were held at background levels since water contamination would be negligible (Figure 5-1). Figures 5-6 and 5-7 illustrate the shelf ecosystem simulation and Figures 5-8 and 5-9 illustrate a typical offshore location within the contaminated area.

The impact of the sediment contamination is felt by the benthos in the ecosystem model, with the pelagic system affected only by food chain transfer through predation on the benthos (Figures 5-6 and 5-9). PCB concentrations in the benthos and demersal fish increase by more than an order of magnitude in both the shelf (Figure 5-6) and the offshore (Figure 5-8) case while plankton remain uncontaminated. On the shelf, pelagic fish consume some benthos (eg. shrimp) and demersal fish, and so become slightly more contaminated with PCBs. Offshore, the pelagic and benthic systems are relatively uncoupled and so the contaminated is confined to the benthos. In both cases, the

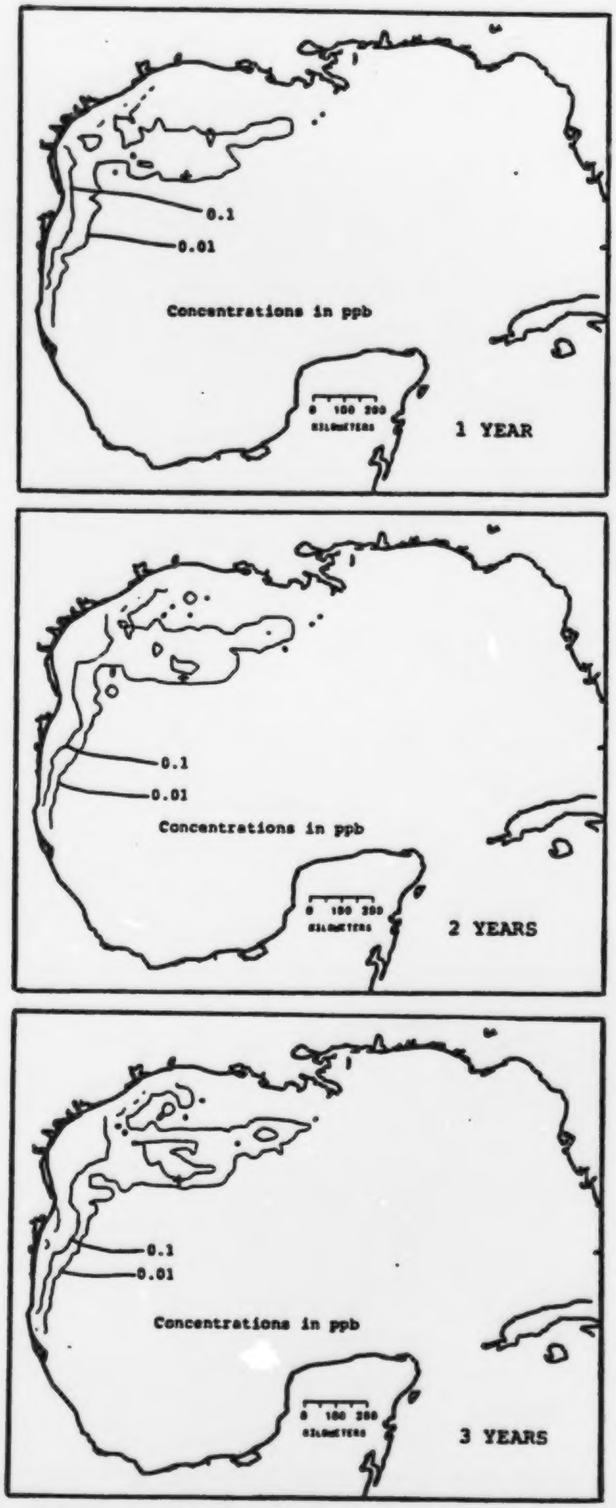

Figure 5-3 Accumulation of PCB on ocean floor due to continuous 26 day incineration cycle, source strength of 1 kg/hr, microlayer residence time of 1 week. a) End of 1 year, b) end of 2 years, c) end of 3 years.

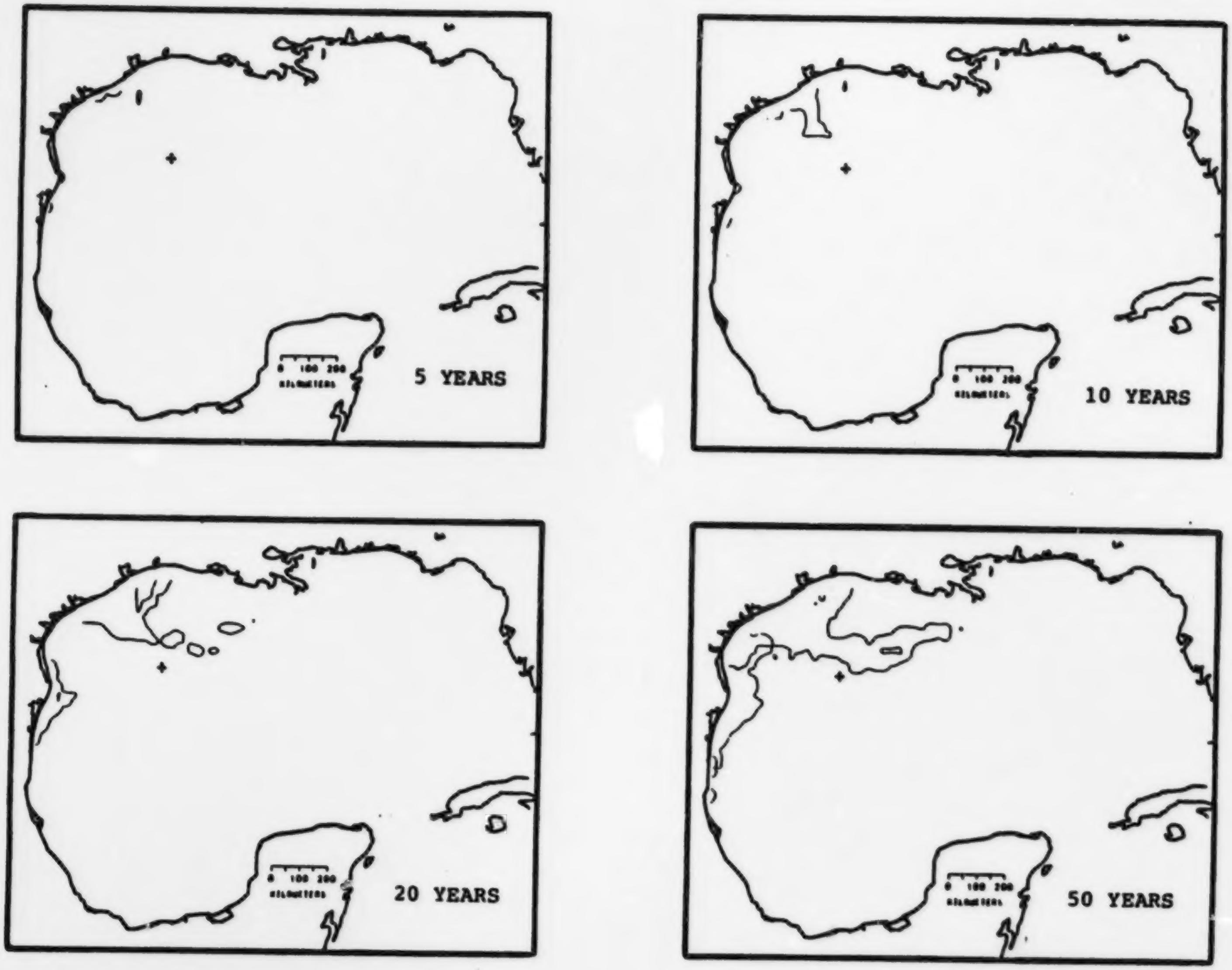

Figure 5-4 Area within 1 ppb PCB contour on ocean floor due to continuous 26 day incineration cycle, source strength of 1 kg/hr, at the end of a) 5 years, b) 10 years, c) 20 years, d) 50 years.

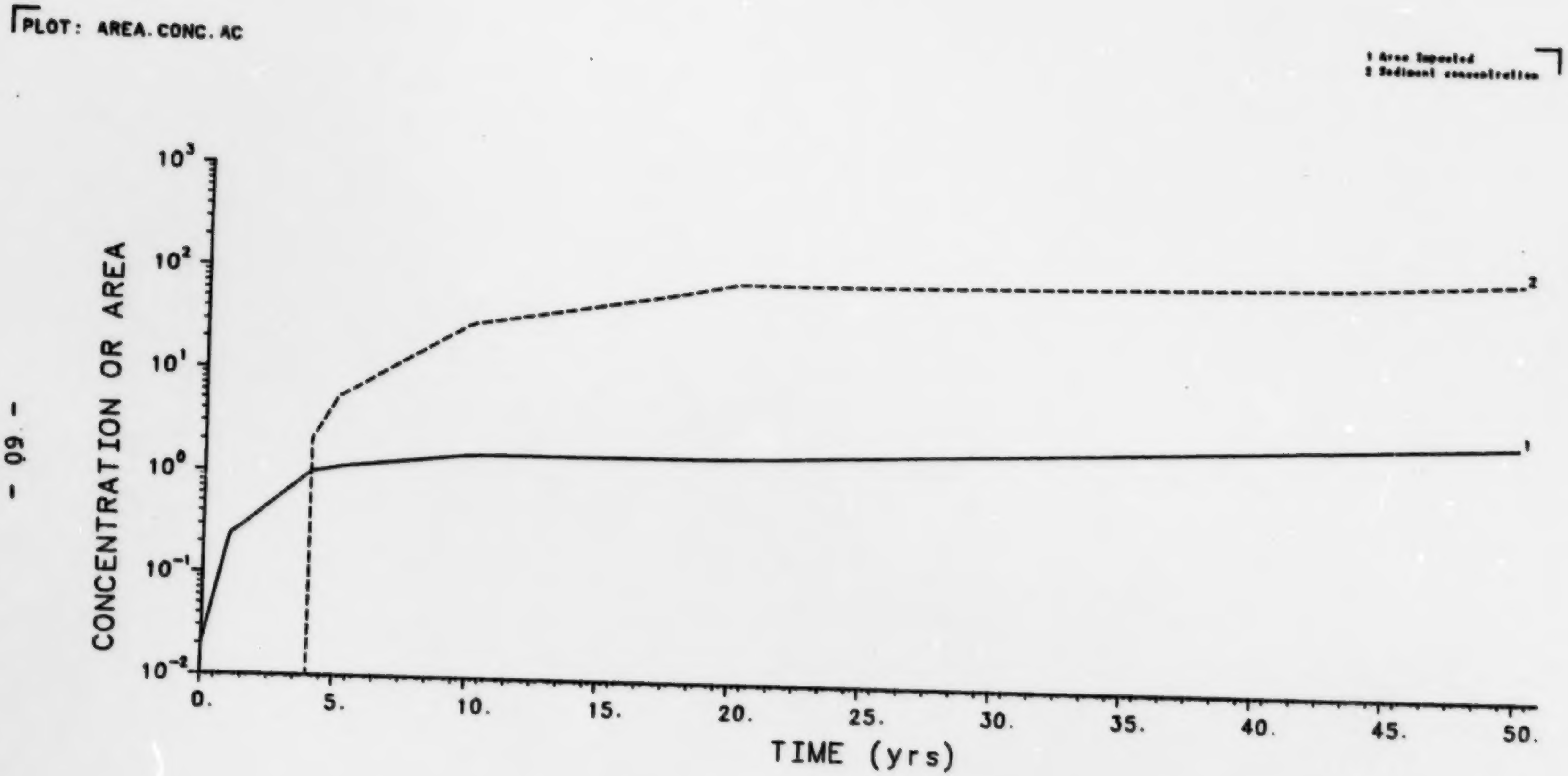

Figure 5-5 Average PCB concentrations (ppb) in the sediments within the 1 ppb contour and the area within the 1 ppb contour (10^5 km^2) as a function of time since initiation of burning every 26 days for 11 days each burning cycle.

PLOT: BIOM.CONC.S5

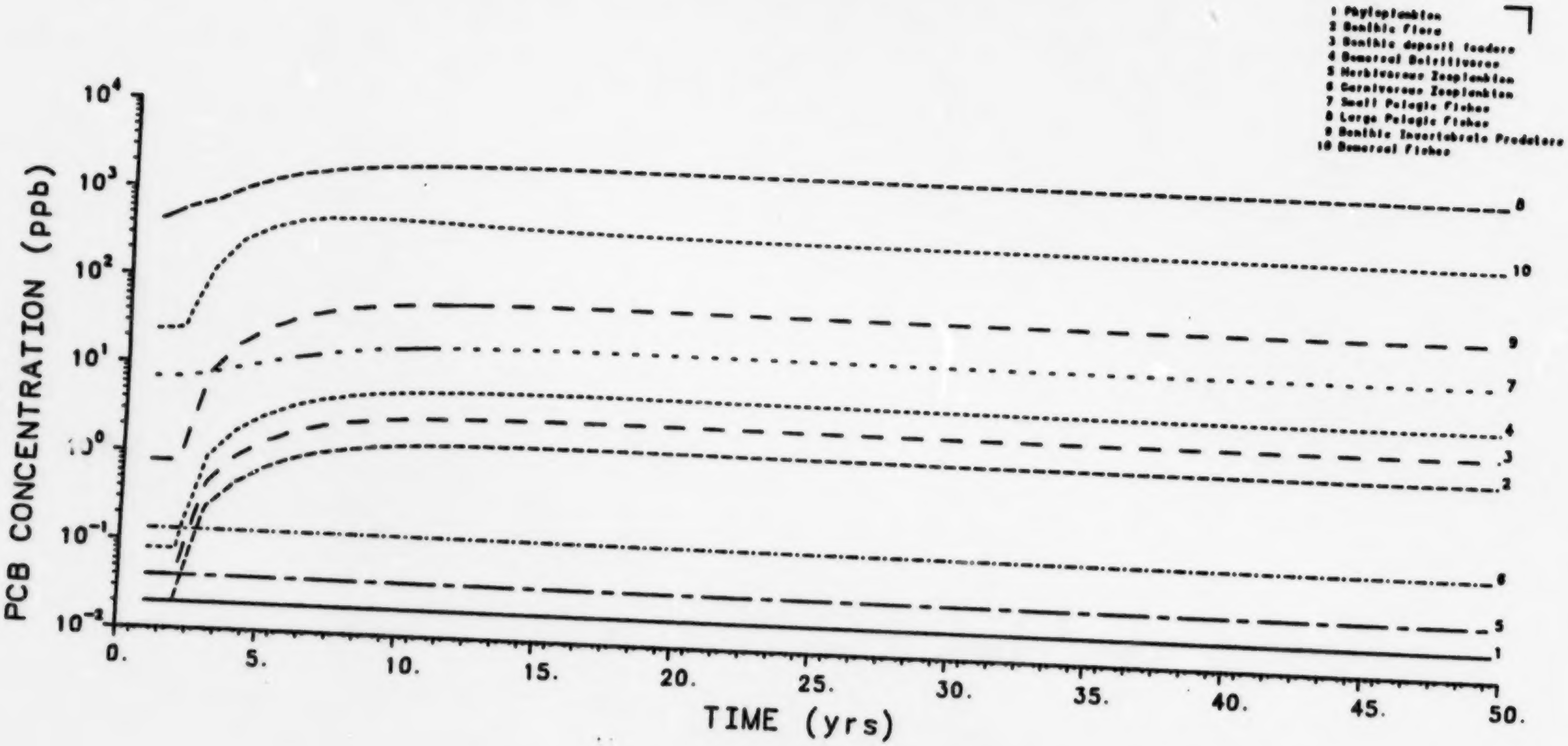

Figure 5-6 PCB concentrations in shelf ecosystem compartments following initiation of burning at 2 years every 26 days for 11 days. The exposure levels used here are approximately 70 times greater than those resulting from the expected incineration efficiency of 0,999999.

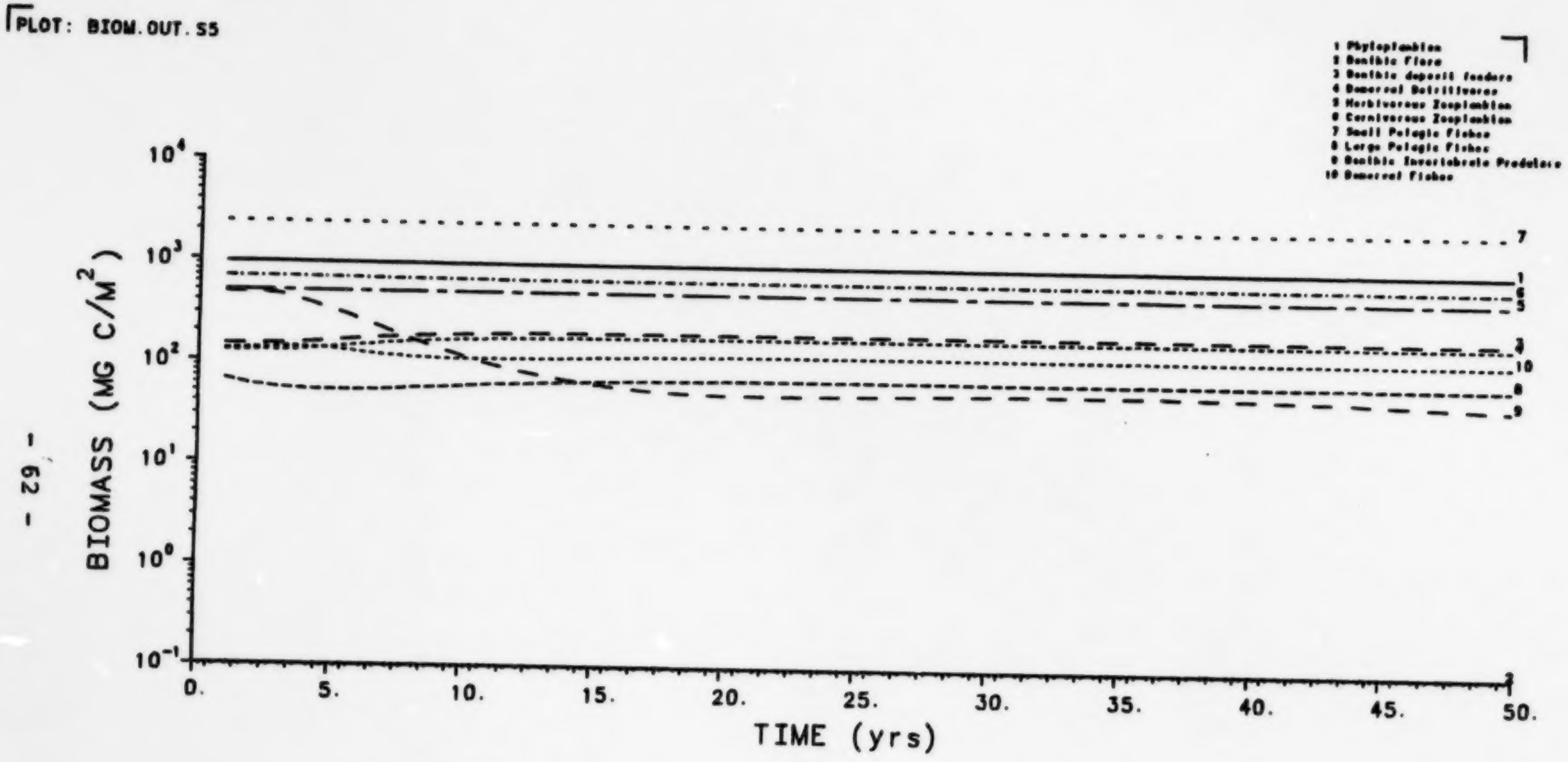

Figure 5-7 Biomass levels of shelf ecosystem compartments following initiation of burning at 2 years.

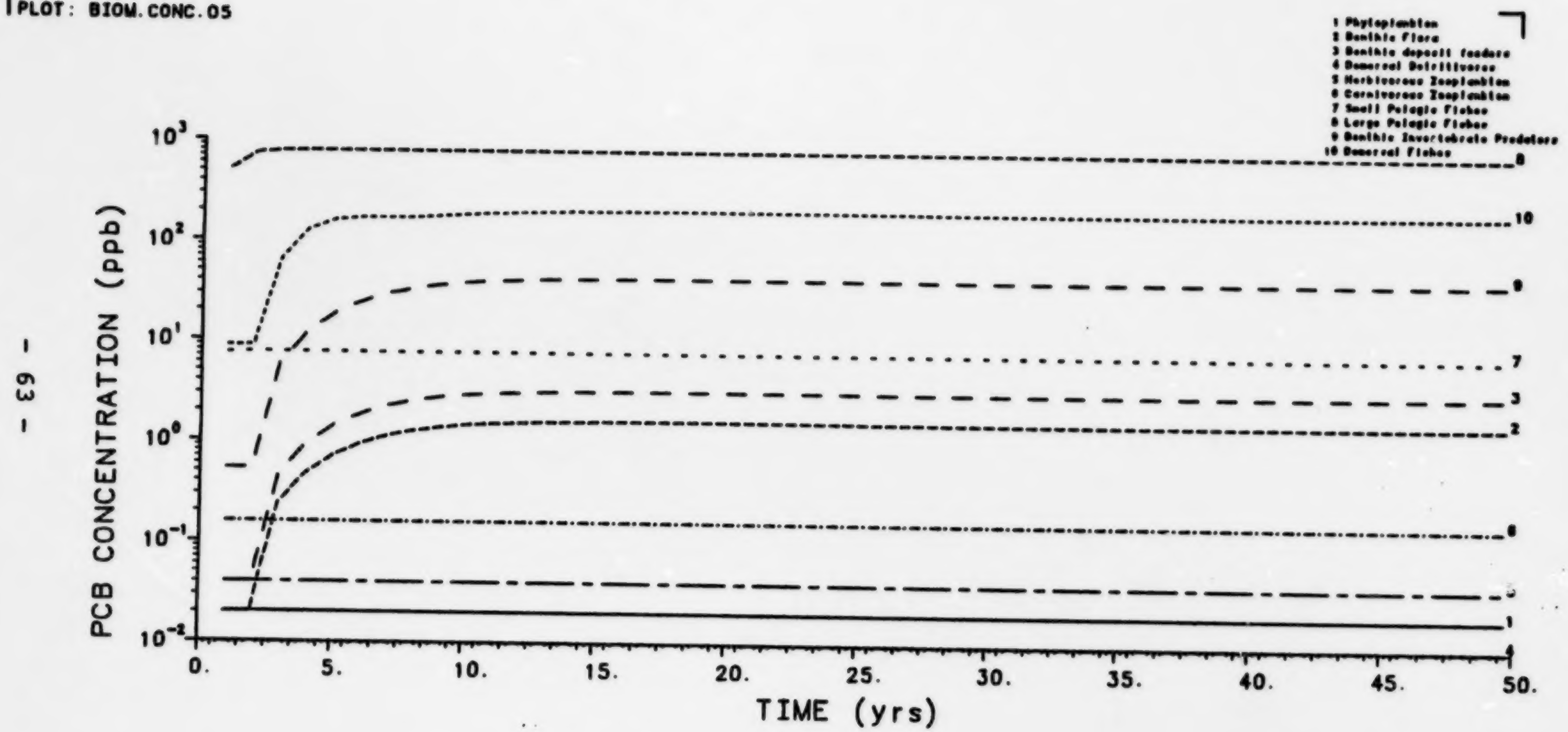

Figure 5-8 PCB concentrations in offshore ecosystem compartments following initiation of burning at 2 years every 26 days for 11 days.

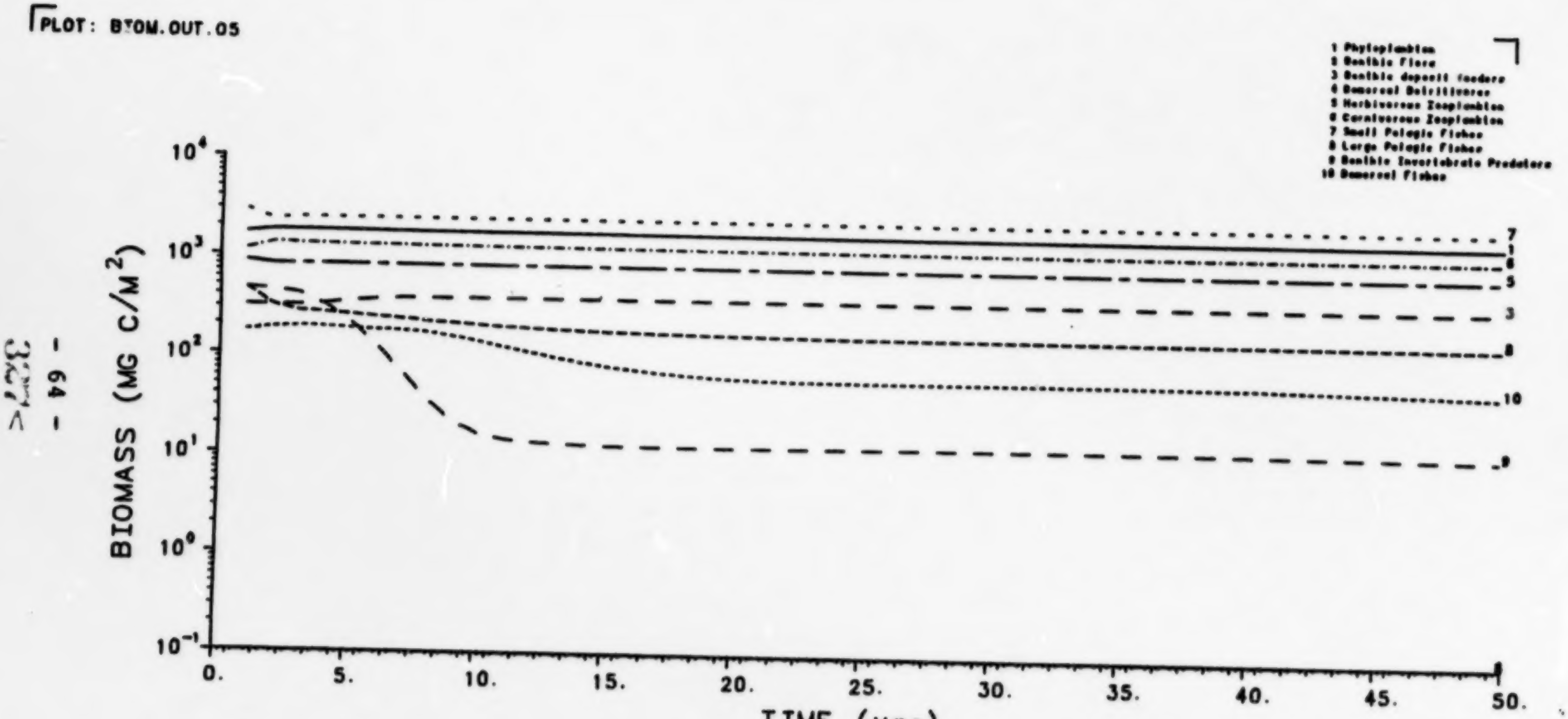

Figure 5-9 Biomass levels of offshore ecosystem compartments following initiation of burning at 2 years.

route of PCBs into the ecosystem is through consumption of contaminated sediments by benthic deposit feeders (ecosystem compartment #3) and demersal detritivores such as shrimp (compartment #4) in the shelf case.

Biomass levels of plankton and pelagic fish are relatively unchanged in the model simulations (Figures 5-7 and 5-9). However, benthic invertebrate predators decline in biomass due to the toxicity of the PCBs in the sediments. Mortality of the benthic deposit feeders is also increased due to PCB-toxicity. However this compartment (#3) does not decline but actually increases slightly, since the decline in their predators releases the predation pressure on them. Lower prey concentrations cause a decline in demersal fish, particularly in the offshore case and subsequently a slight decline in large pelagic fish which prey, in part, upon them.

The biomass changes in Figures 5-7 and 5-9 are based on the assumption that all of the PCBs in the sediment are available to the benthos. Since PCBs sorb onto particles, all the PCB present in the sediment may not be toxic. Laboratory toxicity tests are generally performed in clean water where all the PCB is available to the organisms. Thus Figure 5-7 and 5-9 represent a worst-case scenario for the 1 kg/hr emission rate and probably overestimate biomass effects. No measurable effects are expected from emissions from the Vulcanus, which are about 200 times less than those assumed in the modelling. It would be desirable in the future to consider bioavailability of PCBs in the environment and its relationship to mortality, as well as sublethal effects of PCBs on organisms and the ecosytem.

The atmospheric concentration of PCBs in the Gulf area is about 0.05 ng/m^3, whereas in the mid-seventies it was as high as 0.5 mg/m^3. Using a dry deposition rate of 8 m/hr yields an estimate of background flux to Gulf waters from dry deposition of about 3.5 to 35 g/km^2 yr. Wet deposition rates probably would be similar. Given the Gulf of Mexico surface area of about 1.5×10^6 km^2, the net input of PCBs to the Gulf from the atmosphere has been in the range of about 10 to 100 MT/yr.

By contrast the deposition of PCBs from the Vulcanus, averaged over the approximately 90,000 km^2 area affected by the plume, is about 0.1 g/km^2 yr. This flux rate would represent an increase over background of about 0.15 to 1.5 percent. The total amount of PCBs deposited into the Gulf from the Vulcanus of about 0.009 MT/yr would represent an increase over background flux to the entire gulf of about 0.01 to 0.1 percent.

Metals

The proposed burning of toxic wastes in the Gulf of Mexico will introduce metals into the atmosphere in the exhaust plume. Some of these metals (Cu, Ni, Se and Cr) are essential elements, i.e. required in small quantities by living organisms. However, all metals, whether essential or non-essential, (e.g. As, Cd, Pb, Ag and Hg which are present in the wastes) are toxic in higher concentrations than those which the organisms are adapted to. Therefore, it is necessary to determine whether metal concentrations resulting from plume fallout are within the concentration range which is nontoxic to the organisms involved.

Metals are highly reactive and exist mainly in complexed form in seawater, rather than as free metal ions. Metal complexes include dissolved inorganic complexes; colloidal metal oxides (e.g. Fe and Mn oxides) and metal precipitates; organic complexes with dissolved organic ligands (chelates including amino acids, fatty acids, lipids, humic acids, polysaccharides, proteins, etc.), and metals sorbed onto inorganic particles, organic detritus and live and dead organisms (Duedall et al, 1983). Determining the concentrations of these metal species in seawater requires knowledge of all metal concentrations, salinity, pH, organic ligand composition and concentrations, and particulate composition and concentrations. A few chemical models have been attempted, but poorly defined natural seawater systems cause difficulties in model performance (Morel et al, 1975; Morel et al, 1979; Jackson and Morgan, 1978).

The high reactivity of metals makes them useful to organisms as catalysts, which is how most metal utilization occurs. Metals are incorporated into enzymes where they may be involved in acid/base conversions (e.g. Zn) or in redox (oxidation/reduction) reactions (e.g. Fe). Their reactivity requires that metal transport, storage and control of intracellular concentrations be mediated by specific binding proteins. Excesses of metals and pollutant metals can interfere with these mechanisms and enzyme activity. Extracellular metal can interfere with membrane transport (George, 1982). Thus unicellular organisms such as phytoplankton, protozoans, bacteria and invertebrate and vertebrate eggs are particularly vulnerable.

Plants and animals have evolved detoxification mechanisms which involve sequestering on structural proteins and formation of metalic granules, which are stored intra- and/or extracellularly (George, 1982; Bascom, 1982). Bascom has argued that these natural defenses are efficient enough to detoxify animals in relatively high "toxic" concentrations of pollutant metals, and metal toxicity effects should not normally be a problem.

Detoxification mechanisms prevent bioaccumulation and biomagnification up the food chain, and thus lethal and sublethal effects are limited for the most part to toxicity via water and food. Measurements of metal concentrations in marine organisms do not show increases with increasing trophic level, as do organics such as PCBs, T and organic (as opposed to inorganic) mercury (Fowler, 1982; Bascom, 1982).

Several recent investigations have shown that toxicity to phytoplankton (Manahan and Smith, 1973; Sunda and Guillard, 1976; Anderson and Moral, 1978; Jackson and Morgan, 1978; Huntsman and Sunda, 1980; Sunda and Huntsman, 1983), crustaceans (Andrew et al, 1977; Sunda et al, 1978), mollusks (Stephenson and Taylor, 1975) and fish (Sprague, 1968; Zitko et al, 1973; Brown et al, 1974; Kramer and Neidhart, 1975) is related to the free metal ion activity (or concentration) and not related to the total metal concentration. Change in organic or inorganic ligand concentration, with no change in total metal concentration, results in a change in degree of toxicity. In addition, the relative free metal concentrations of other metals can increase or decrease toxicity of the metal in question (Huntsman and Sunda, 1980; Sunda and Huntsman, 1983).

Several investigations of the toxicity to phytoplankton of Cu free ion activity have indicated that free Cu activities in natural seawater are within the range found to be toxic to some species of algae (Sunda and Guillard, 1976; Anderson and Morel, 1978; Jackson and Morgan, 1978; Huntsman and Sunds, 1980). Thus small increases in total copper without increased complexation could be toxic to many species of phytoplankton. This could be of particular significance in open ocean areas where organic ligand and particulate concentrations are very low (Sunda and Guillard, 1976). Since phytoplankton are unlimited by a number of metals only within a narrow range of free metal ion concentrations, metals other than Cu may also be at or near toxic concentrations in open ocean seawater (Huntsman and Sunda, 1980). Thus small changes in metal ion activities could result in decreased productivity and change in phytoplankton species composition, and thus food chain structure.

Assumed concentrations of metals in the wastes to be burned are 100 ppm for As, Cd, Cr, and Ni. Thus, amounts in the entire cargo are 4×10^5g, for each metal.

This amount will be emitted in the plume over an incineration period of 1 to 2 weeks. The subsequent distribution in the atmosphere is discussed in Appendix G. The metals will to a large degree be complexed as they fall out from the atmosphere in the form of oxides or adsorbed to particulates emitted in the plume. Metals entering the sea surface may largely be adsorbed to organics in the microlayer. The 100 um sea surface film is greatly enriched in organic matter and particulates relative to the bulk water concentration below (Garrett, 1965; Harvey, 1966; Williams, 1967; Duce et al, 1972; Liss, 1975). Thus, it is likely that much of the metal fallout will be rendered non-toxic by complexation. A change in free metal concentration in the water can only be predicted by a detailed chemical model of metal speciation. Knowledge of toxicity levels of free metal concentrations is incomplete, making assessment of biological effects difficult. Removal rates of metals via sedimentation is also poorly quantified, but is likely to be rapid, given the high affinity of metals for particles and the high flux rates of particulates to the deep sea (Edler and Fowler, 1977). The effect of this increased metal load on the benthos should then be quantified. A metals activity model component is suggested for future work.

The coupled atmospheric plume-ocean transport model indicates that the oceanic surface area affected will be on the order of 90,000 km^2 after a period of one or more years of burning.

As described in Appendix G, a "generic metal" has been defined for modeling purposes. This substance has the mean specific gravity, particle diameter, and molecular weight of arsenic, cadmium, chromiun, and nickel (Table 5-1). The coupled atmospheric oceanic transport model shows a mean flux of a metal contained in the waste at 100 ppm to be about 30 g/km^2-yr. Thus about 60% of the metals released from the stack will be deposited in the ocean before the plume reaches shore. According to the atmospheric model, after a single 11 day burn of wastes, 60% of the metals in the cargo will be deposited over an area of 7500 km^2. The direction and location of the plume will vary by season so that a much larger area will be affected after multiple burns (See Appendix G). Assuming 60% of the metals in the cargo are deposited within this area, three assumptions are examined with respect to the depth of penetration of the metals: (1) metals are retained in the microlayer (100 um thick); (2) metals are mixed to a depth of 1 meter; (3) metals are mixed throughout a 60 m mixed layer. If the metals were completely retained in these layers, the resulting concentrations would be as indicated in Table 5-2. Background concentrations for the surface open-ocean are included for comparison. Concentrations of metals in the microlayer range from 1 to 50 times surface-water background levels (Piotrowicz et al, 1972, Duce et al, 1972). Under this worst case scenario, only cadmium inputs will exceed mean ambient levels if mixed to more than 1 m, and all metals will exceed ambient levels if penetration is only into the microlayer. These calculations assume that all the metal is retained within the layer for the 11 days of the burn, an unrealistic assumption for the microlayer and 1 m depths of penetration, but probably realistic for the 60m surface mixed layer. Thus, the 60 m case is a reasonable worst-case prediction. Toxicity will be dependent on the degree of

Table 5-2. Concentrations (ng/l = parts per trillion) of selected metals found naturally in oceanic surface water (background) and depositional inputs after the incineration of one cargo load assuming penetration into the 100 um microlayer, to 1 m depth and to the bottom of a 60 m mixed layer. Background levels are from Quinby-Hunt and Tarekian (1983).

Metal	Surface 60 m Background	Penetration Depth Microlayer	1 m layer	60 m layer
As	1100	320×10^3	32.0	0.53
Cd	0.3	320×10^3	32.0	0.53
Cr	268	320×10^3	32.0	0.53
Ni	146	320×10^3	32.0	0.53

complexation and adsorption these metals undergo. Since the microlayer contains elevated concentrations of organics and particulate material, it is likely that much of the excess metal will be complexed and non-toxic. Metals mixing to 1 m or 60 m depth must pass through the microlayer, thus increasing the chances for complexation. If adsorbed onto particles, excess metals could be rapidly transported to depths where they are likely to be dispersed to undetectable levels, partly by dilution and partly by the fact that Cd and Ni concentrations generally increase with depth.

HCl

The burning of PCBs and EDC will release considerable quantities of chlorine, which will enter the sea surface as HCl, hydrochloric acid. EDC, for example is 72% chlorine by weight. If a 3800 MT load is 50% EDC, then about 1370 MT of chlorine will be released from the stack during a 6 day (25 MT/hr) burn operation. As an extreme example, we can hypothesize that this entire amount is deposited within a 14 km radius of the ship. Field measurements in the North Sea (Weitkamp et al. 1984) suggest a typical atmospheric lifetime of HCl on the order of 1 hour, or 50% loss to the sea surface within 14 km of a source in a 4 m/sec wind. Actual lifetime is, not surprisingly, extremely weather-dependent, with faster losses to the ocean during precipitation events. Assuming as a worst case 100% loss from atmosphere to ocean within a 14 km radius of the ship implies a deposition density of about 2 grams chlorine per square meter, if ocean advection relative to the ship is neglected. Typical wind speeds in the Gulf of Mexico are 5-10 m/sec, so we have chosen a relatively low value which is unlikely to persist over a full 6 day period.

Seawater has a large buffering capacity, mainly due to the CO_2-carbonate system (Riley and Chester, 1971), such that acids and bases are readily absorbed, keeping pH generally constant and near 8.0. A measure of the ability of seawater to absorb acid is the alkalinity. Alkalinity is defined as the total concentration of ions capable of adsorbing protons (H^+, acid). In practice, the only contributing ions in significant quantity in seawater are HCO_3^-, CO_3^{-2} and $B(OH)_4^-$ (bicarbonate, carbonate and borate, respectively). The alkalinity of open ocean seawater (salinity = 35%) is 2.38 milli-equivalents (mequiv.) per liter (Riley and Chester, 1971). This is equivalent to 2.38 mmole HCl/l.

Seawater at the ocean surface has a pH of about 8.2. As acid is added to seawater, pH decreases very slowly, only to about 7.5, after 1 mequiv./l of acid has been added. A second mequiv./l of acid will reduce the pH to 4.0. Further additions of acid reduce pH to 2.0. (M.E.Q. Pilson, University of Rhode Island, pers. comm.). Thus surface seawater could adsorb up to about 1 mmole HCl/l before pH would change significantly. Effects on organisms and the ecosystem would not be expected with acid inputs below this level.

The amount of acid input per volume depends on the depth to which the acid is mixed after falling on the sea surface. Table

Table 5-3. Input of HCl to the ocean as a result of burning 3800 MT of waste containing 50% EDC. Analysis assumes complete deposition within 14 km of the ship.

Depth of Mixing	HCl Input After 6 days (mmole/l)	HCl Input After 1 day (mmole/l)
100 um	564.0	94.0
1 cm	5.64	0.94
6 cm	0.94	0.16
1 m	0.056	0.009

5-3 gives input rates assuming various depths of penetration from the surface microlayer (100 um) to 1 meter. If the surface layer is not renewed over the 6 day period (i.e. dead calm, no waves, no vertical density currents), the acid input would be absorbed with no significant change in Ph if it were mixed down to greater than 6 cm. If renewal of the surface occurred at least daily (100% turnover of the surface layer each day), the acid input would be absorbed if mixed down at least 1 cm. Given the normal turbulence of the upper 2-3 meters of the sea surface as the result of wind and waves, penetration of the acid to >6 cm is extremely likely. Moreover, renewal of the sea surface layer would be frequent enough to mix the acid down into the water column where it would be absorbed with no noticeable effect.

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6. Marine Ecosystem Effects Due to Accidental Releases

6.1 PCB Spills

6.1.1 Ecosystem Model Predictions at Background Levels of PCBs

Reported background levels of PCBs in oceanic waters range from non-detectable to 35 ng/l (Harvey et al. 1973). From a survey of the North Atlantic surface waters (<200 m), Harvey et al. reported an average concentration of 20 ng/l (= 0.02 ppb). This level was assumed as background for the Gulf of Mexico for both the water and the sediments. Phytoplankton (ecosystem compartment #1) and benthic flora (ecosystem model compartment #2) are assumed to be in equilibrium with the surrounding medium and therefore contain 0.02 ppb PCB.

Table 6-1a gives steady state PCB levels by trophic compartment assuming the indicated assimilation and depuration rates. PCB concentrations increase up the food chain in all simulations. Regional differences are not significant at lower trophic levels, but increase further up the food chain depending on the turnover rates of the populations involved. Thus demersal fish are less contaminated as water depth increases and their productivity decreases.

For the Gulf of Mexico, reported PCB levels in fish range up to 530 ppb (Goldberg, 1975, Gadbois and Maney, 1983). Boehm and Hirtzer (1982) measured 10-90 ppb PCB and Gadbois and Maney (1983) report 20-200 ppb PCB in Gulf demersal fish (muscle), which compares favorably to the range in the simulations of

Table 6-1a. Predicted PCB levels (ppb) by compartment and site when water and sediment concentrations are held at a background level of 0.02 ppb PCB.

#	Compartment	Asimi-lation Rate	Depuration Rate (Day^{-1})	Mobile Bay	Shelf	Offshore
3	benthic deposit feeders	.20	.10	.04	.04	.04
4	demersal detritivores	.20	.05	.07	.07	--
5	herbivorous zooplankton	.20	.10	.04	.04	.04
6	carnivorous zooplankton	.20	.05	.15	.13	.16
7	small pelagic fish	.60	.01	2.9	6.7	9.2
8	large pelagic fish	.60	.005	--	750	850
9	benthic invertebrate predators	.40	.03	.40	.75	.50
10	demersal fish	.60	.01	100	25	10

10-100 ppb. Gadbois and Maney measured 50-340 ppb PCB (average 170 ppb) in small pelagic fishes including menhaden from Mobile Bay. Goldberg (1975) gives 1-10 ppb PCB as the range typical of pelagic fish in the open North Atlantic. The model predicts 2.9-9.2 ppb in small pelagic fish. Simulated PCB levels in large pelagic fish are 750-850 ppb, which may be compared with 90-400 ppb in tuna from the Mediterranean Sea (Boehm and Hirtzer, 1982).

6.1.2 Mobile Bay Spill Scenarios

Estimating Mobile Bay In-Stream PCB Concentrations

Mobile Bay is a semi-enclosed estuarine system with currents resulting primarily from tidal action, large freshwater river inflow, and wind-driven flows (Raney and Youngblood 1982). A shipping channel 122 m (400 ft) wide and 12.3 m (40 ft.) deep runs the 50 km length of the bay. The average depth over the 900 square kilometer plan area is only about 3 meters (U.S. Army 1983).

We will investigate three release quantities for floating PCB waste spills. The release quantities will be 8 tanks, (the total load), two tanks, and one half tank. The total load of the incinerator ship, 4000 MT, is taken to be 35% PCBs, the remainder being some more volatile non-hazardous substance to enhance combustion.

Spill conditions in each case will be assumed the same:

-spill site at point x (Figure 6-1)
-wind out of the north or southwest at 5 m/sec
   (most probable events, Raney and Youngblood 1982)
-spill occurring at mean low water
-peak tidal velocities of about 0.6 m/sec

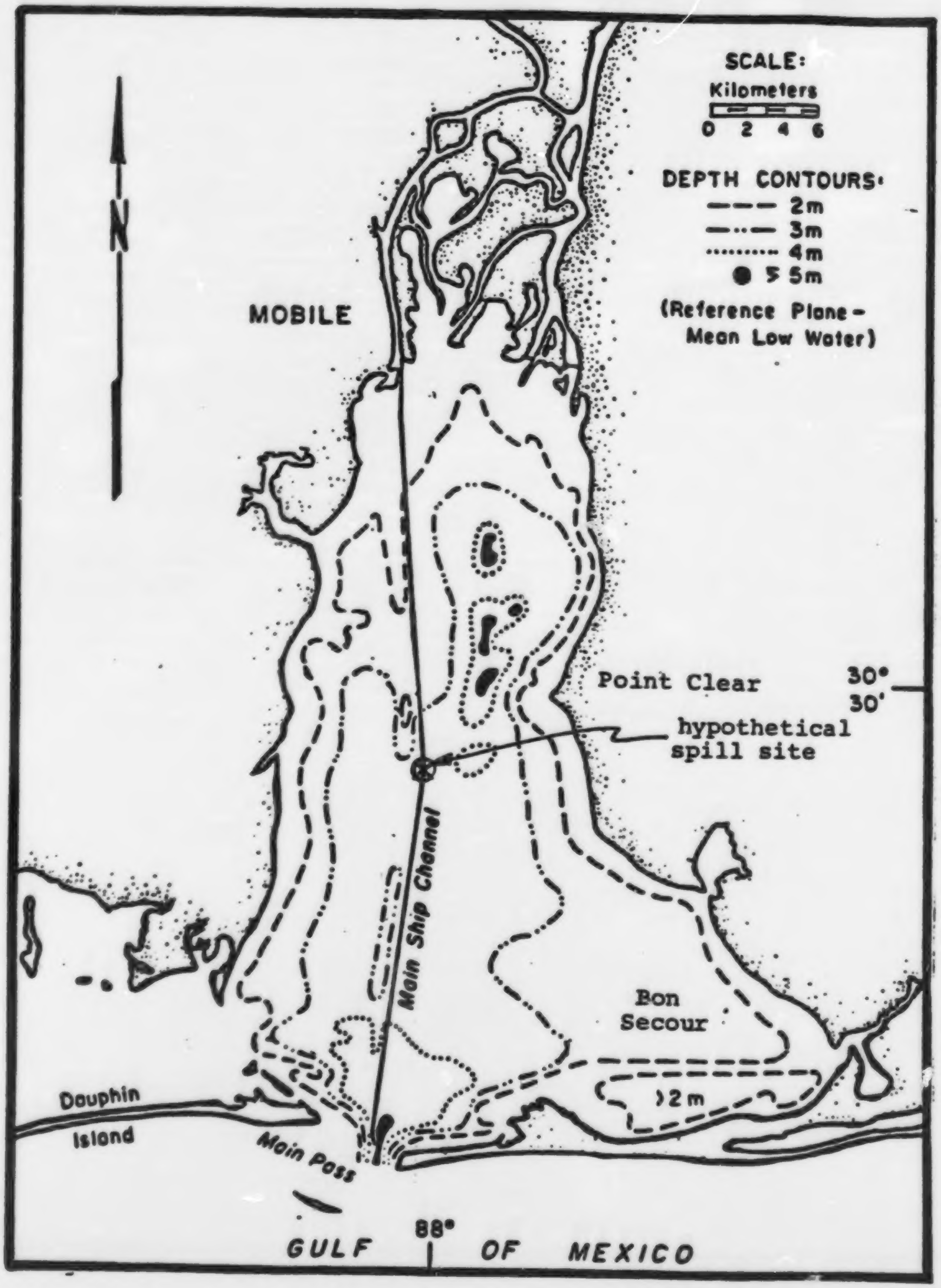


Figure 6-1. Mobile Bay, showing assumed spill location.

-a 24 hour tidal cycle
-wind driven surface advection at 3% of wind speed.

It will also be assumed that the entire quantity spilled is released within the first few hours of the scenario, and that no spill containment or cleanup activities are undertaken at any time.

Floating Waste Scenarios

If the PCBs are dissolved in a light fuel oil or other non-hazardous substance such that the net specific gravity of the waste is less than that of Mobile Bay water, the waste would probably behave much like an oil spill. Spreading curves for the three spill sizes of interest have been computed based on the oil spill spreading work of Hoult (1972) and are shown in Figure 6-2. The estimated spill trajectories for the two most probable wind speeds and directions (Raney and Youngblood 1982) are shown in Figure 6-3a for spills occurring at low tide. The spill trajectories for waste releases at high tide are shown in Figure 6-3b.

Depending on the amount of waste released (8, 2, or 1/2 tank spills) and the time required to reach shore, the length of shoreline initially impacted and the total swept area can be estimated from Figure 6-2. Total swept area is of interest since the high suspended solids concentration typical of Mobile Bay, combined with evaporation of the lighter molecular weight fractions of the waste (see Section 6.3), will result in a net transfer of PCB from the surface slick to the Bay floor along the trajectory. Estimates of swept area and length of coastline impacted are given in Table 6-1b.

PCB waste arriving at the shoreline will probably cause the immediate local area to become abiotic through a combination of smothering and toxic action. For fine sediment areas typical of Mobile Bay, the waste will gradually penetrate into the sediment, and will probably remain for long periods of time. This conclusion is based on observations of the Amoco Cadiz and other oil spills (Gundlach et al. 1985), and observations of PCB

Case A: north wind, spill at low tide
Case B: southwestwind, spill at low tide
Case C: north wind, spill at high tide
Case D: southwest wind, spill at high tide

	Case A	Case B	Case C	Case D
8 Tanks				
Coastal length (km) Swept Area (km^2)	7 140	3 30	6 120	4 60
2 Tanks				
Coastal length (km) Swept Area (km^2)	4 80	2 20	3 60	3 45
1/2 Tank				
Coastal length (km) Swept Area (km^2)	2 40	1 10	2 40	1 15

Table 6-Ib. Estimated swept area and length of coastline initially impacted for each of four scenarios shown in Figures 6-3 and 6-4.

Table 6-1c. Ecosystem exposure parameters for three spills of floating 35% PCB waste in Mobile Bay.

Spill Size	Percent Seafloor Coverage	PCB Mass* on Bottom (mt)	Mean Concentration in Contaminated Zone (ppm)
8 tanks (4000 mt)	15%	700	61
2 tanks	9%	175	25
1/2 tank	4.5%	44	13

*initially

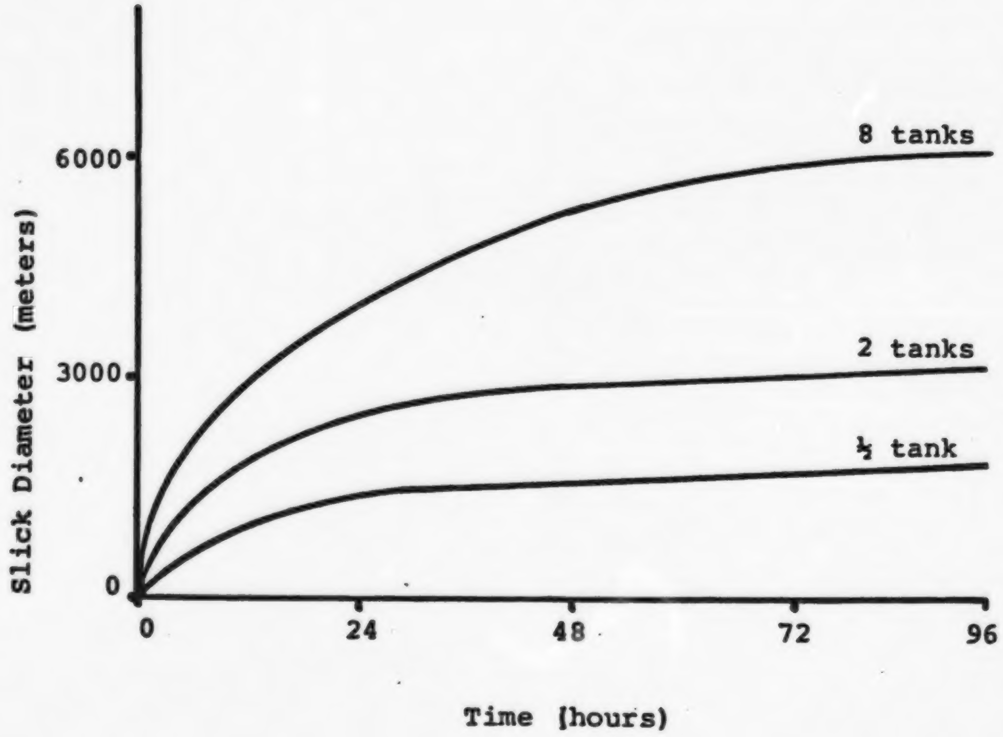


Figure 6-2. Estimated spreading curves for the three spill sizes of interest.

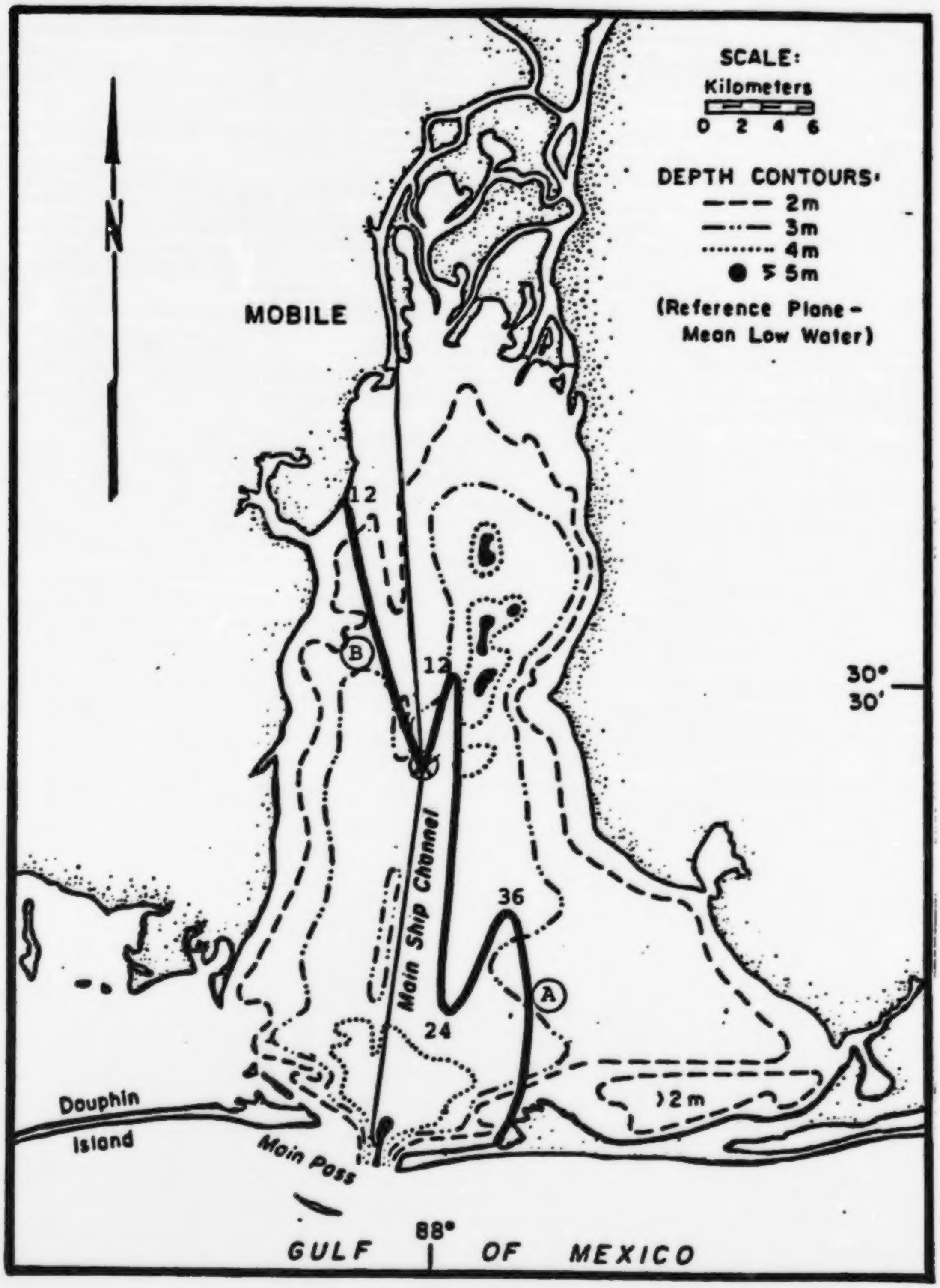


Figure 6-3a

Estimated trajectories for a spill occurring at low tide. A: north wind; B: southwest wind. Numbers along trajectories indicate elapsed time in hours.

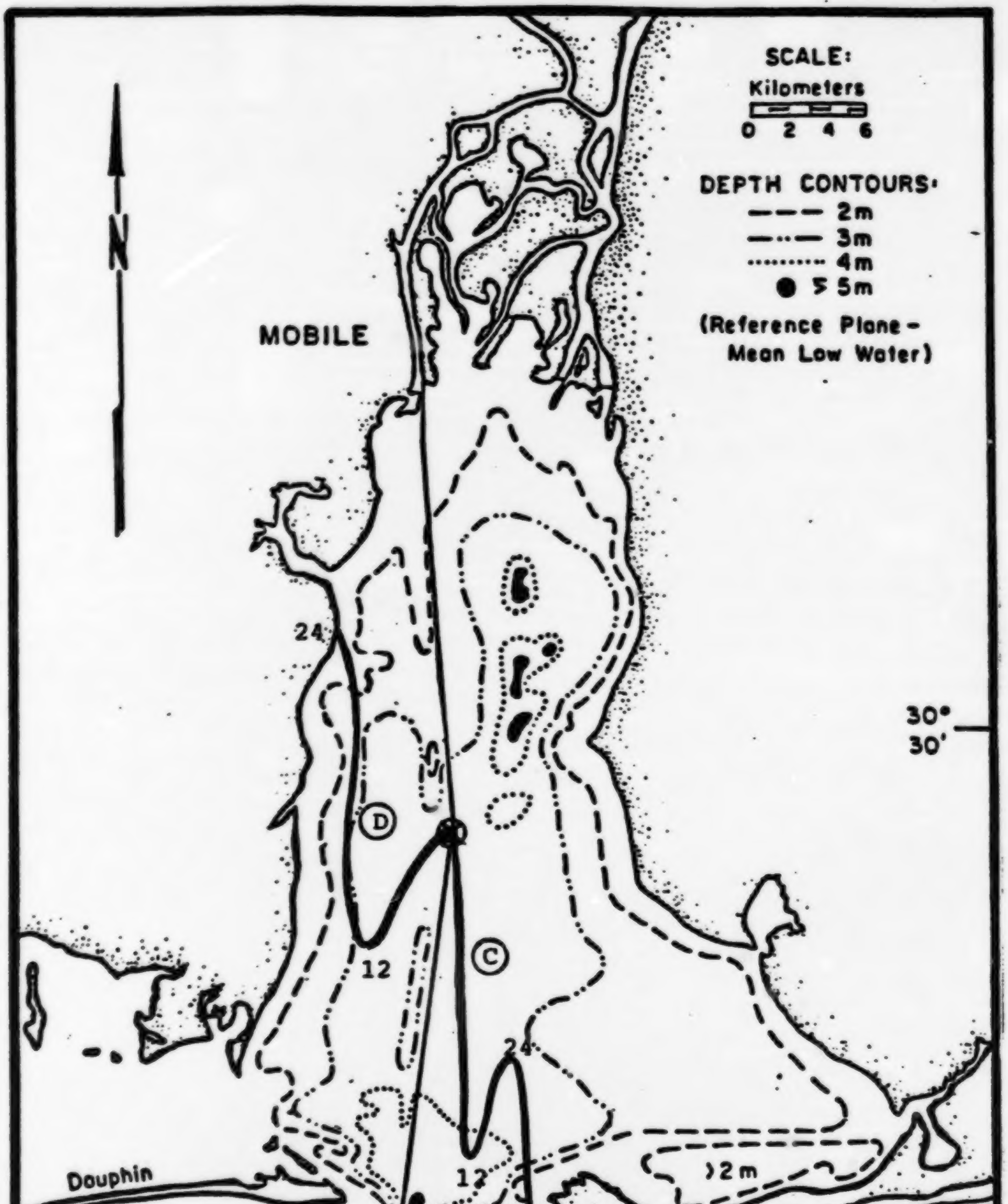
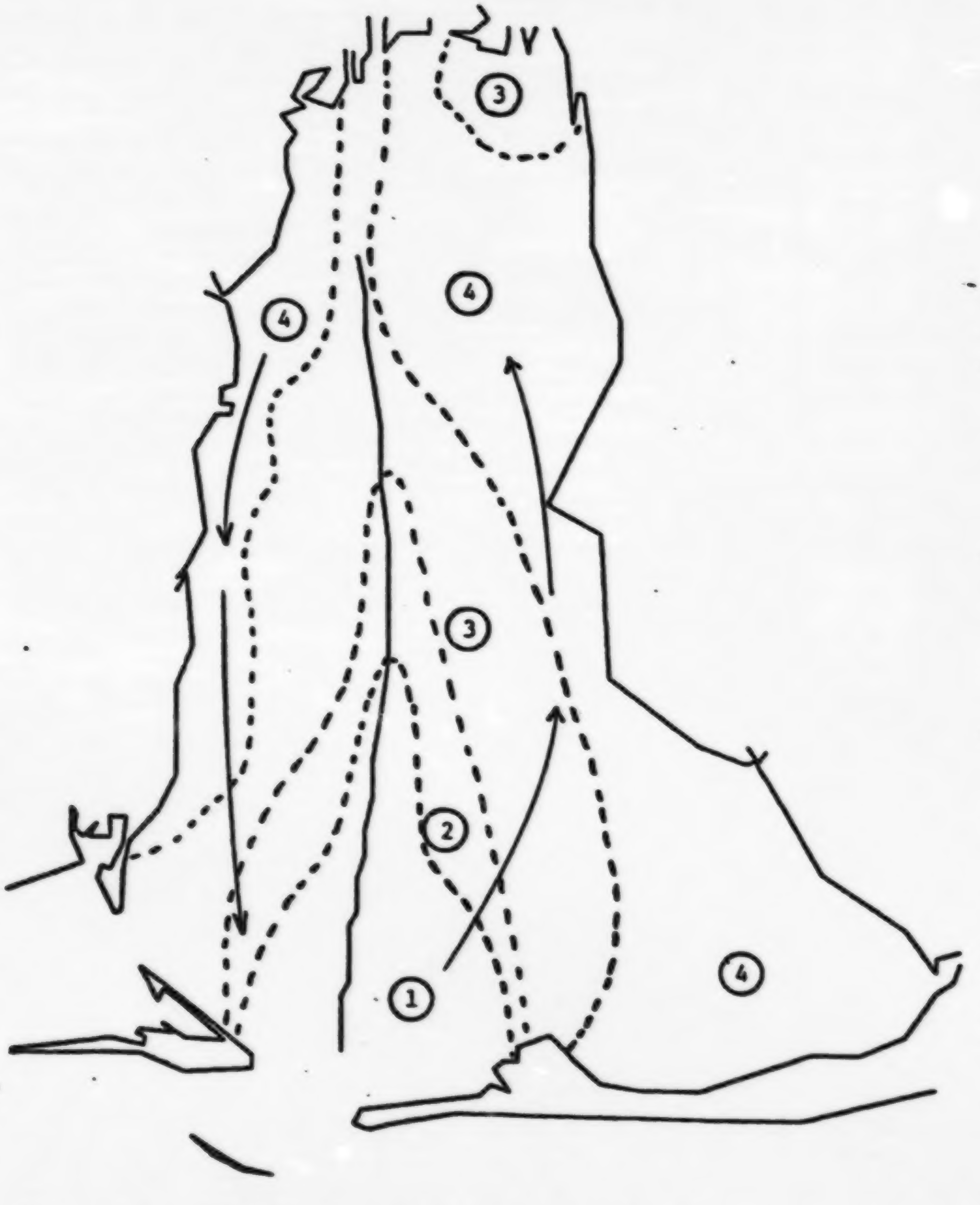


Figure 6-3b
Estimated trajectories for a spill occurring at high tide. C: north wind; D: southwest wind. Numbers along trajectories indicate elapsed time in hours.

immediate local area to become abiotic through a combination of smothering and toxic action. For fine sediment areas typical of Mobile Bay, the waste will gradually penetrate into the sediment, and will probably remain for long periods of time. This conclusion is based on observations of the Amoco Cadiz and other oil spills (Gundlach et al. 1985), and observations of PCB behavior in New Bedford Harbor in Massachusetts (Farrington 1982).

PCBs "raining out" onto the Bay floor as the slick passes overhead will be more or less susceptible to resuspension and redistribution, depending on local hydrodynamic energy (Figure 6-3c). Regions of low hydrodynamic energy are much more susceptible to sediment deposition than are regions of higher average energy levels. Erosion rather than deposition will occur in regions of very high average hydrodynamic energy levels. Figure 6-3c indicates that Bon Secour and the region above Point Clear are possible regions for deposition. Sediment disturbed from the channel apron on the west side of the channel appears to have a much greater chance of being flushed from the bay than does sediment from the channel apron on the east side of the channel, where spill trajectories A and C (Figures 6-3a and 6-3b terminate). Sediment disturbed from the east side of the channel appears to have a good chance of being deposited in Bon Secour or in the upper eastern part of the bay because of low average hydrodynamic energy levels in these regions (Raney and Youngblood 1982). PCBs associated with these sediments would be transported in the same pattern. Measurable levels of PCB transport through



Region 1: high hydrodynamic energy level
Region 2: moderately high hydrodynamic energy level
Region 3: medium hydrodynamic energy level
Region 4: low hydrodynamic energy level

⟶ mean circulation direction

Figure 6-3c. Average hydrodynamic energy levels in Mobile Bay over a tidal cycle (from Raney and Youngblood 1982)

sediment resuspension in protected estuaries can occur on time scales of a few years (Farrington 1982).

An additional but probably less important source of PCB redistribution will be through biotic uptake and active swimming. The net effect in the long term (20-50 years) would be to spread PCBs over larger portions of the Bay than were initially impacted.

Response of Mobile Bay Ecosystem to PCB Spills

For an oil spill in an estuary, initial oil distribution can be approximately partitioned as follows (Gundlach et al. 1985).

- evaporation: 30% - 50%
- shoreline stranding: 40% - 60%
- sedimentation: 10% - 20%
- dissolution: 5% - 10%
- biodegradation/photooxidation: 1% - 2%

The PCB waste under consideration will lose about 50% of the lower boiling point fractions within 48 hours of release (Section 6-3). These fractions do not include the PCBs themselves, which reside in the boiling point cuts above 554 degrees F (Table 6-3). PCBs are also highly resistant to biodegradation and dissolution. A corrected mass balance for the PCB portion of the waste might therefore be:

- evaporation: 1% - 5%
- shoreline stranding: 30% - 70%
- sedimentation: 30% - 70%
- dissolution: 0% - 1%
- biodegradation/photooxidation: 0% - 1%

Clearly the partitioning between shoreline stranding and sedimentation is of major interest here. In general, the longer the time between spill start and land fall, the more sedimentation will have occurred. Of the four cases investigated (Figures 6-3a and 6-3b), case A (Figure 6-3a and Table 6-1b) represents the maximum potential for initial dispersal of PCBs over the Bay. We will therefore focus on this "worst case" for ecosystem impact estimates.

Figure 6-3c shows the net circulation direction and regions of the Bay most susceptible to sediment resuspension and deposition. Figure 6-3d shows an estimate of the bay floor area initially receiving PCB from the slick passing overhead. The figure includes the effect of surface slick spreading, and an assumed settling velocity for PCB particulates from 1.0 to 0.1 cm/sec. For a 3 meter depth, most particulates will settle within 1 km of the slick edge. For smaller spills, the initially impacted area will scale approximately as shown in Table 6-1b. From Figure 6-3c, it is evident that much of the spill deposition area near the mouth of Mobile Bay will be susceptible to resuspension. PCBs deposited in this region will be transported northward and eastward on flood tides, and will be removed from the bay on ebb tides. Without performing a more thorough analysis it is not possible to reliably quantify the long term dynamic transport in more detail. For purposes of ecosystems impact estimation, the following assumptions have therefore been made:

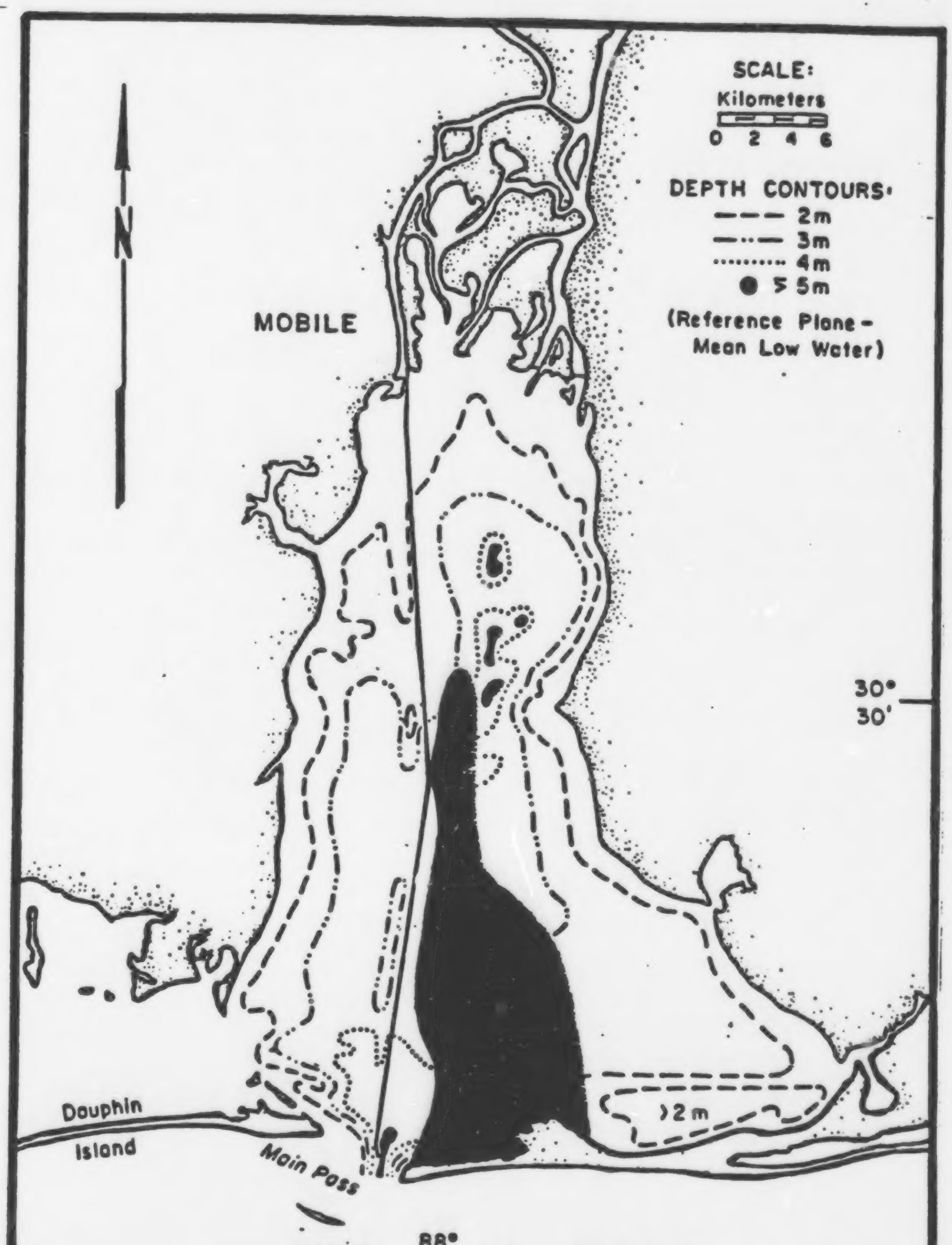


Figure 6-3d

Approximate area of initial PCB deposition for Case A (see Table 6-1b and Figure 6-3a, 8 tank spill).

(1) 50% of PCBs in the spill reach the bay floor before the slick strands.

(2) 100% of the PCBs on the bay floor remain in the top 5 cm of sediment, the layer to which the benthic organisms are exposed.

(3) Mobile Bay flushing, at about 12% per day, is high enough to keep mean water column concentrations in the Bay at or near the present background levels.

(4) Through the mechanisms of bioturbation, resuspension and flushing, PCBs will be removed from the top 5 cm of sediments with an e-folding time of 30 years. In other words, the spatially averaged PCB concentration in the top 5 cm of the bay sediments, PCB_5, is given by

$$PCB_5(t) = PCB_5(0) \exp(-0.023t)$$

for t in years, and $PCB_5(0)$ the initial concentration.

Assumption (1) reflects a mean value of the sedimentation range given earlier. Assumption (2) is simply a "worst case" for use in lieu of a more detailed modeling effort. Assumption (3) is based on the fact that 12% flushing per day gives a 97% turnover rate for the entire bay per month. Assumption (4) is relatively arbitrary, but reflects the expectation that Mobile Bay will generally trap about 70% of the suspended sediments in the water column (Ryan and Goodell 1972). Observations in New Bedford Harbor (e.g. Farrington 1982) demonstrate that PCB contamination in estuarine sediments is indeed a long term problem.

With regard to ecosystem exposure, we assume that organisms in Mobile Bay which feed on the bottom will be exposed to PCBs in approximate proportion to the fraction of the bay floor which is contaminated. On the average, then, organisms will be equally likely to feed in contaminated or clean areas. Percent of total bay coverage and mean sediment contamination levels within the affected zone are given in Table 6-1c. The assumption of "averaged feeding" allows for the fact that the percent of the bay floor contaminated will increase with time, while the mean contamination level within that area will decrease.

The ecosystem model responses to spills of 8, 2, or 1/2 tank(s) are shown in Figures 6-4a thru 6-4d. The average benthic exposure levels in the bay immediately following these spills would be 9.2, 2.3, or 0.59 ppm, respectively. These concentrations are all 3-4 orders of magnitude higher than levels lethal to marine organisms (Figure 4-2). Therefore, all three cases show biomass responses similar to that shown in Figure 6-4a. The benthic flora (#2), benthic deposit feeders (#3) and benthic predators (#9) will be largely exterminated from the sediments, with only very low densities of benthic deposit feeders (#3) remaining. As a result, demersal fish will be reduced in biomass due to lowered food resources. Compartment #4, which includes shrimp, increases slightly in biomass in the model run due to lessened predation.

All compartments other than phytoplankton (#1) and herbivorous zooplankton (#5) will be contaminated with PCBs via ingestion of contaminated food (Figure 6-4b, c, d). The main route of the contamination would be through the demersal detritivores (#4), such as shrimp. These animals would ingest contaminated detritus, and are in turn eaten by large (carnivorous) zooplankton (#6) and fishes (#7, 10). The few benthic organisms remaining alive in the benthos after a spill of the entire cargo (Figure 6-4b). If only a portion of the cargo is spilled, the contamination levels are proportionately less

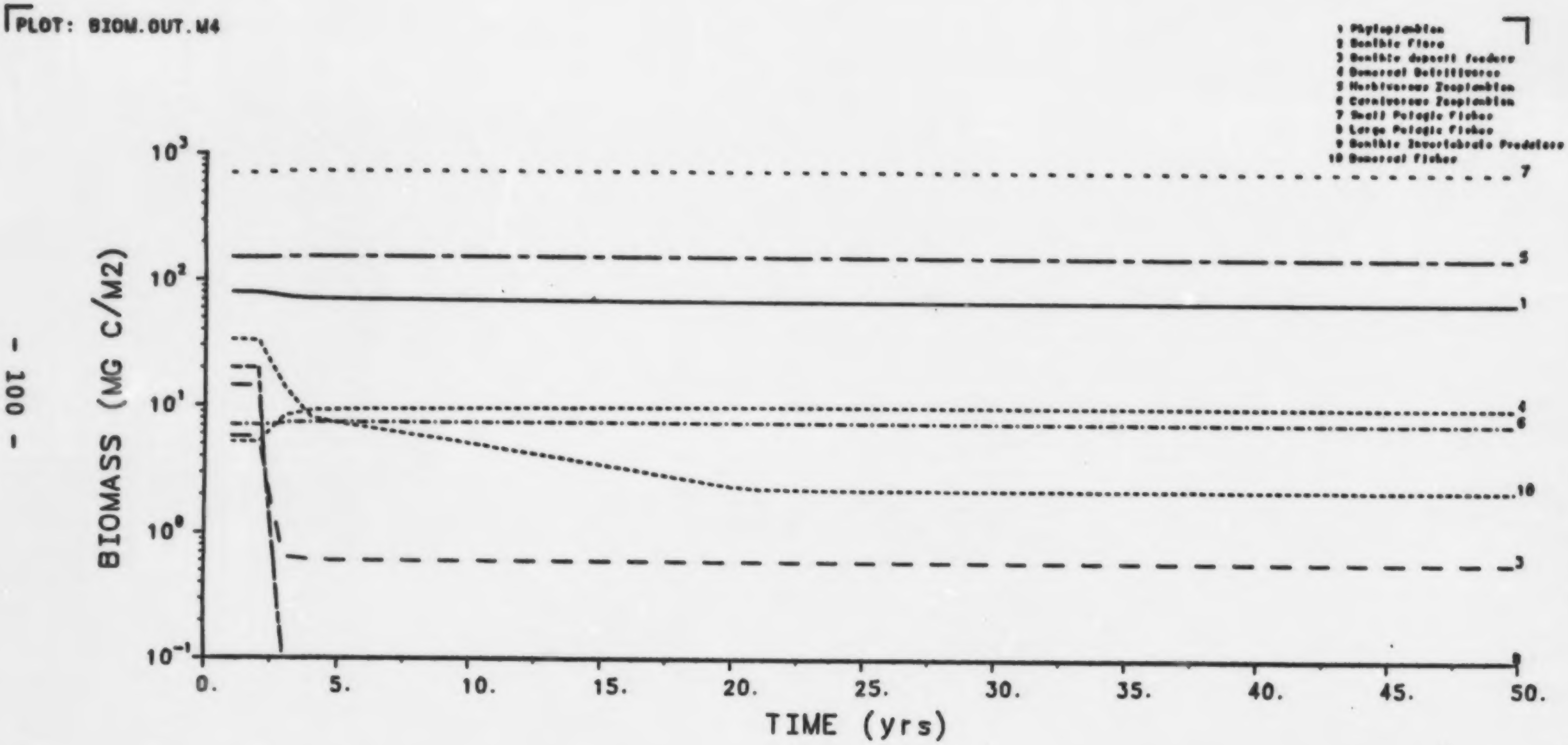


Figure 6-4a. Biomass levels in the Mobile Bay ecosystem before and after a spill of the entire cargo of PCBs occurring at two years. Plotted are annual average biomass levels.

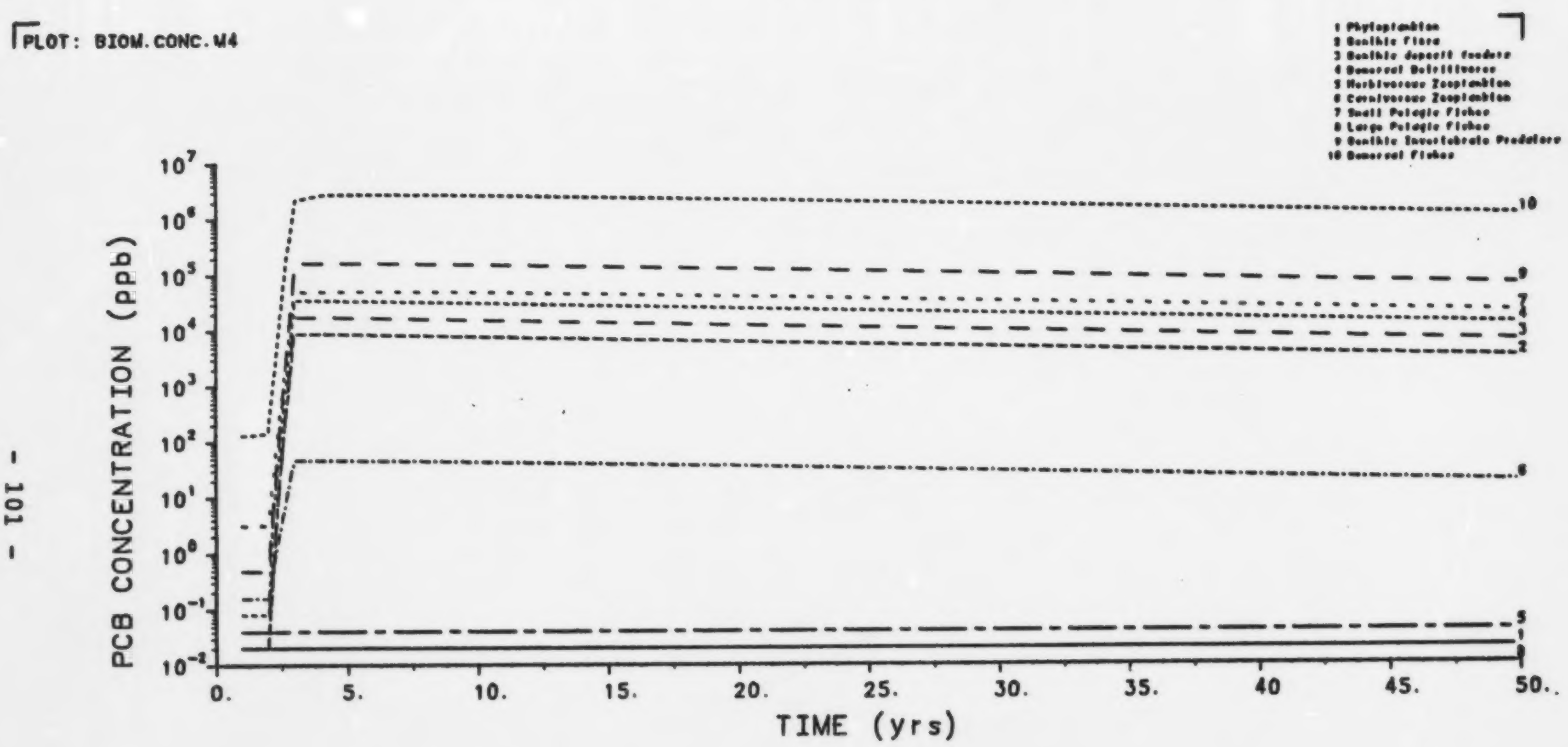


Figure 6-4b. PCB concentrations in trophic compartments of Mobile Bay ecosystem before and after a spill of the entire cargo of PCBs occurring at two years. Plotted are annual average concentrations.

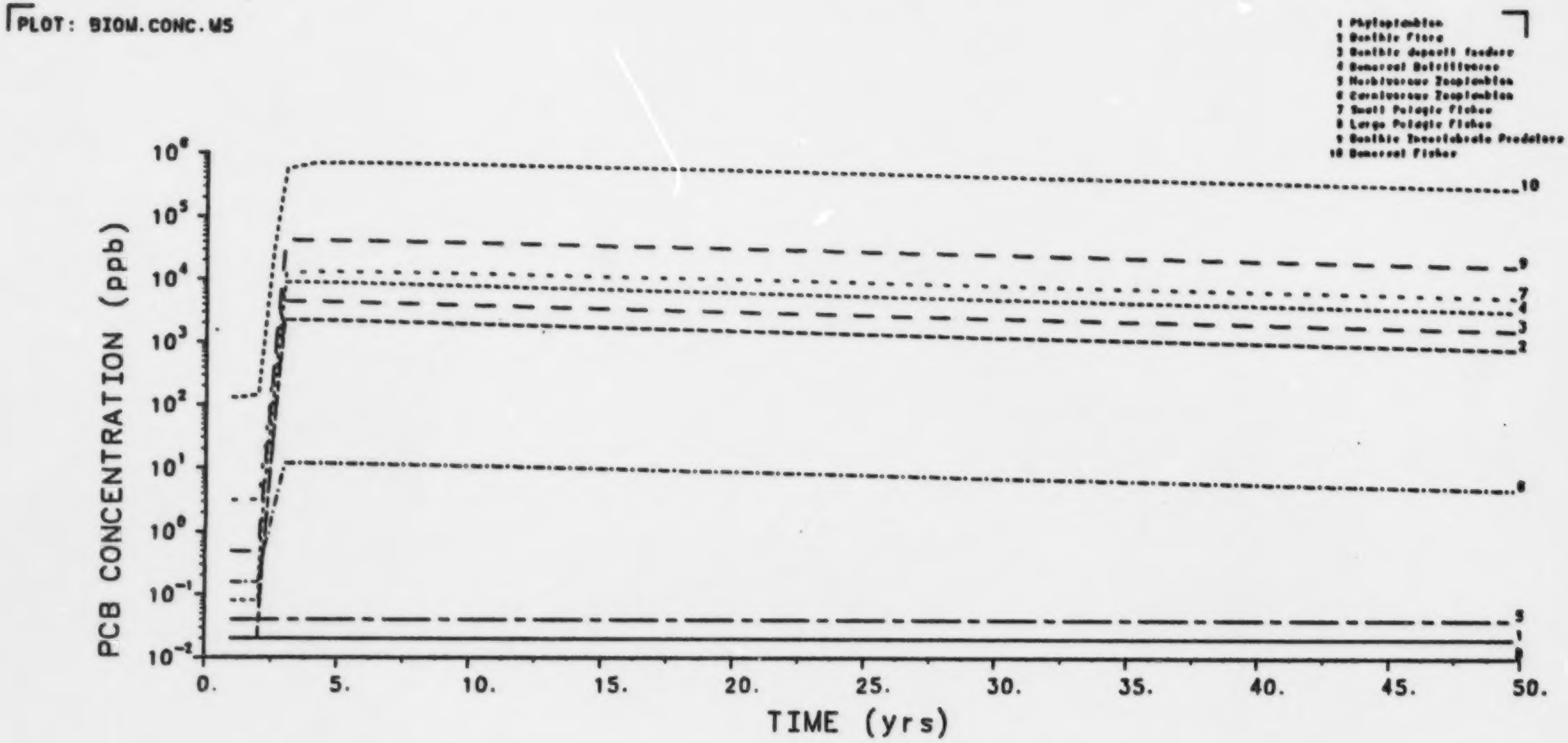


Figure 6-4c. As 6-4b, except only 1/4 of the cargo is spilled.

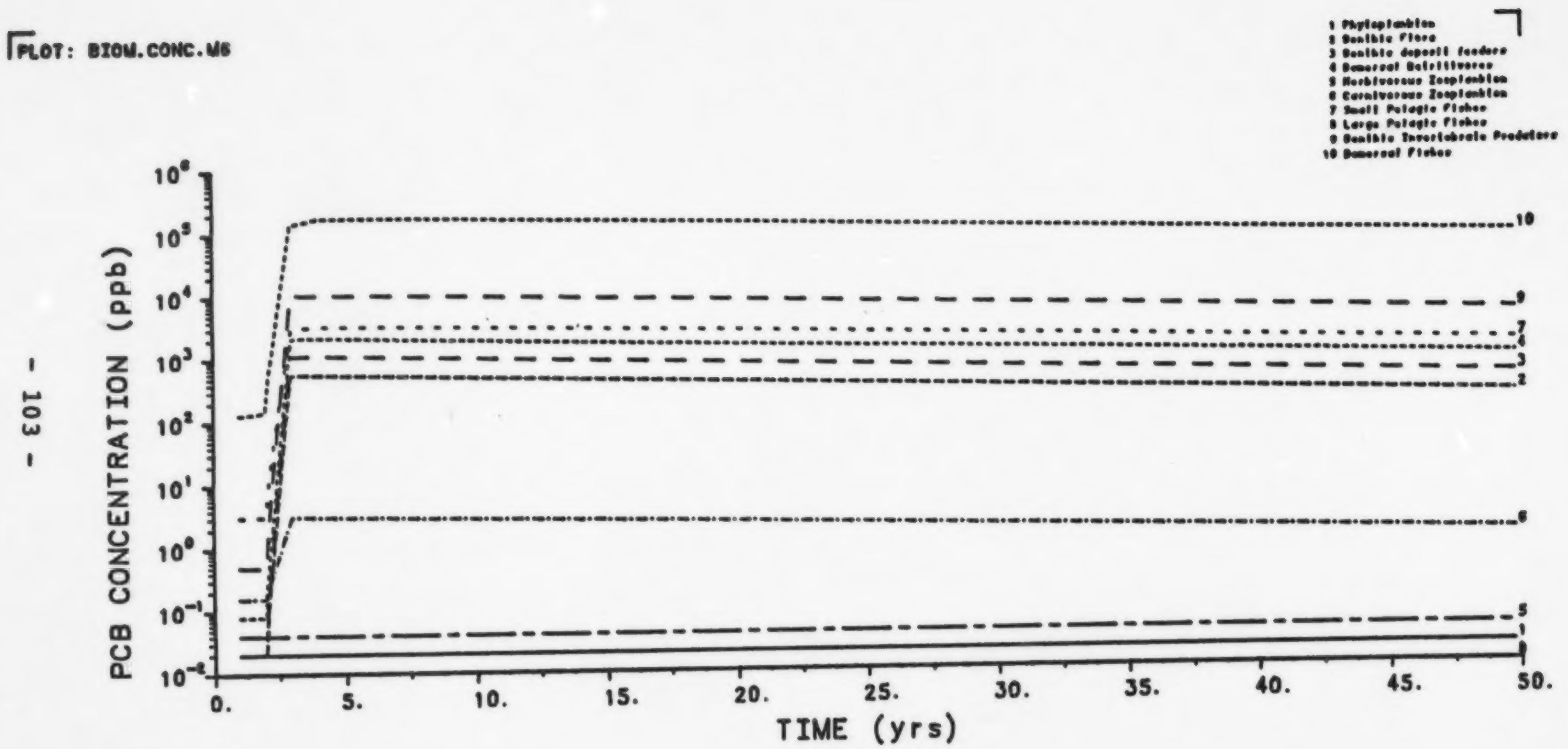


Figure 6-4d. As 6-4b, except only 1/2 tank (1/16 the entire cargo) is spilled.

but, nevertheless, very high (Figure 6-4c, d).

The ecosystem model response described above represents an average response for Mobile Bay as a whole. In actuality there would be areas of the bay more or less contaminated than the average. Thus, the benthos of some areas would probably survive the incident and form a seed population for the more decimated areas. Shallow areas, such as marshes, would be extremely variable in their level of contamination and degree of mortality. It is not known whether predators such as demersal fish would avoid the contaminated areas due to distaste or lowered food supply. Avoidance would serve to lower contamination levels on average.

A number of fish species as well as penaeid shrimp use estuaries and marshes, including Mobile Bay, as spawning and nursery grounds. A spill of PCBs in Mobile Bay could disrupt the grounds and increase mortality of early life stages to a level which would cause a decline in the fishery both inside and outside Mobile Bay. Migrations could also contaminate biota outside the bay as well. The present model analysis has considered Mobile Bay a closed system, where, in actuality, exchanges with the coastal zone occur continuously. The spread of PCB contamination through migration is particularly pertinent to important fisheries such as the shrimp industry.

An additional unknown in the model analysis is the degree to which the PCBs in the sediments are "available" to the benthos in terms of their mortality response. In this analysis, mortality rates of the benthos are a function of the total concentration of

PCBs in the sediments. Some of this material may be effectively removed by sequestering on the sediment. If this is the case, the ecosystem model responses reported here represent worst case scenarios.

6.1.3 Continental Shelf Spill Scenario

Pollutant Transport Model Predictions

The two-layer transport model described in Section 2 was used to simulate an accidental release of the entire cargo of an incineration ship while enroute to the offshore burn zone. The location chosen for the mishap lies along the ship's proposed route, at 29 degrees 45 minutes N, 88 degrees 30 minutes W, where the water depth is less than 100 m. Two scenarios resulting from such a spill were modeled, one in which the entire load of pollutant was assumed to sink, the other in which none of the pollutant sinks. For both scenarios the pollutant release is instantaneous. The surface layer of the model is set at 60 m; below this depth subsurface currents act.

The scenario in which all the pollutant sinks assumes that some of the PCBs will sink quite rapidly to the seafloor, while the remainder would settle at different rates depending on the settling velocities of the particulate matter onto which it adsorbed. Accordingly, three representative settling velocities (0.1, 0.01 and 0.001 m/sec) were selected and uniformly distributed among the PCBs. In other words, 33.3% of the spill sinks at each of these rates, corresponding to mean vertical travel times of 0.28, 2.8, and 28 hours per 100 meters

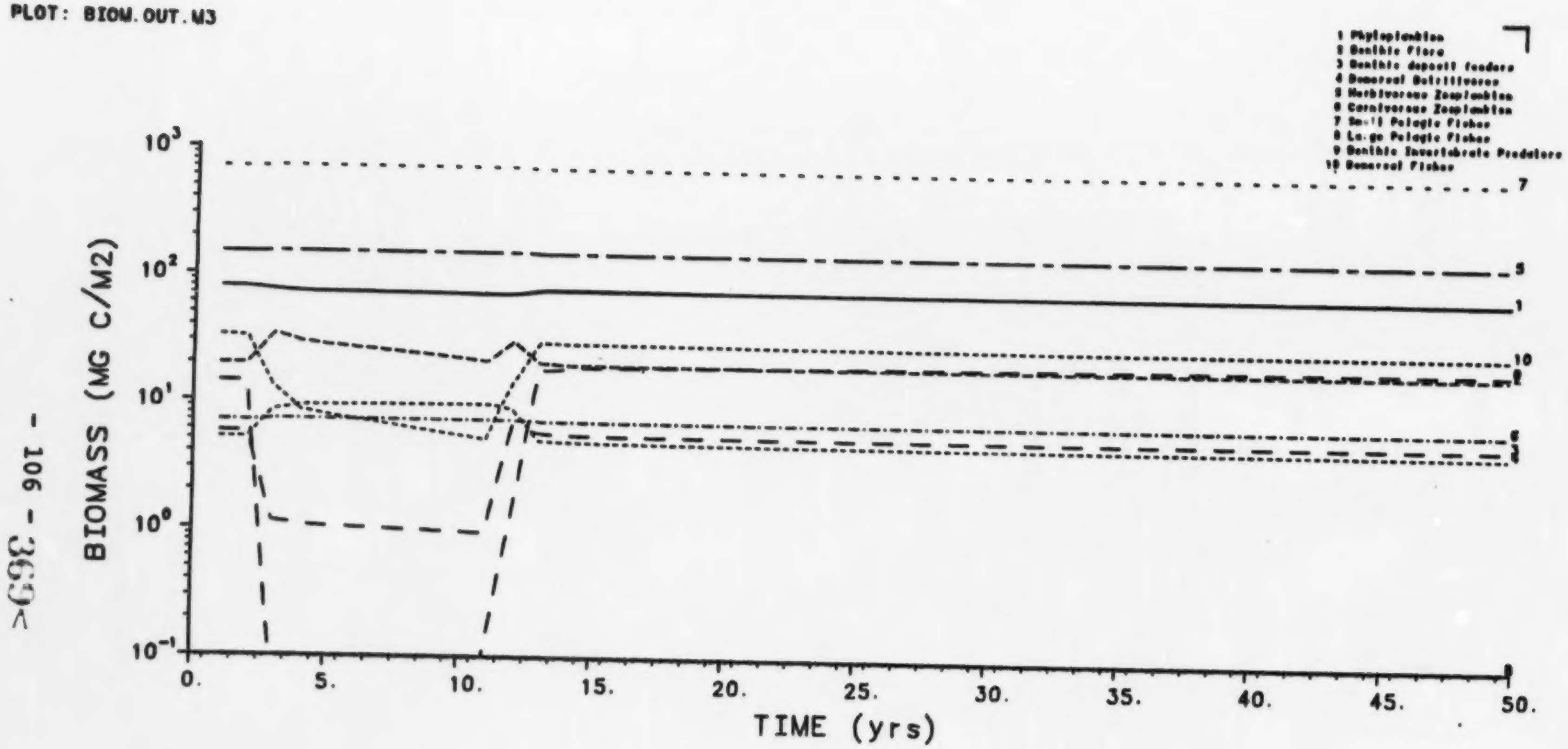


Figure 6-4c. As 6-4b, except spill size is ¼ of the cargo.

respectively. This distribution of settling rates is relatively arbitrary because of lack of detailed information. Due to the relatively shallow depths at the spill location, all of the pollutant reaches the ocean floor within two days. Water column concentrations of PCB are therefore elevated for only a very short period. Bottom concentrations of PCB, however, were very high since only limited dispersion of the pollutant mass occurred during its descent. Resuspension and settling over ensuing years of the bottom PCBs increased the areal extent of the bottom covered and reduced the maximum concentrations. Bottom concentrations of PCB for the scenarios described above are shown in Figures 6-5, 6-6 and 6-7 at the end of 1, 10 and 20 years, respectively. In 6-5a, 6-6a and 6-7a, concentrations following a spill of the entire cargo are shown. If only two tanks (1/4 of the cargo) are spilled, concentrations would be as in Figures 6-5b, 6-6b and 6-7b. The bottom area affected by elevated PCB concentrations for use in the ecosystem effects model is shown in Figure 6-8. For a spill of only 1/4 of the cargo, the affected seafloor area will be approximately the same, since this is a function of the settling rate distribution, not the total mass released. For the smaller spill, concentrations averaged over the seafloor will be proportionately lower.

A second scenario was simulated in which all the pollutant was assumed to float (representative of the behavior of a large oil slick). Although PCBs will tend to settle in the long term, this case was run to provide an estimate of the probable direction of movement and areal extent that would be contaminated

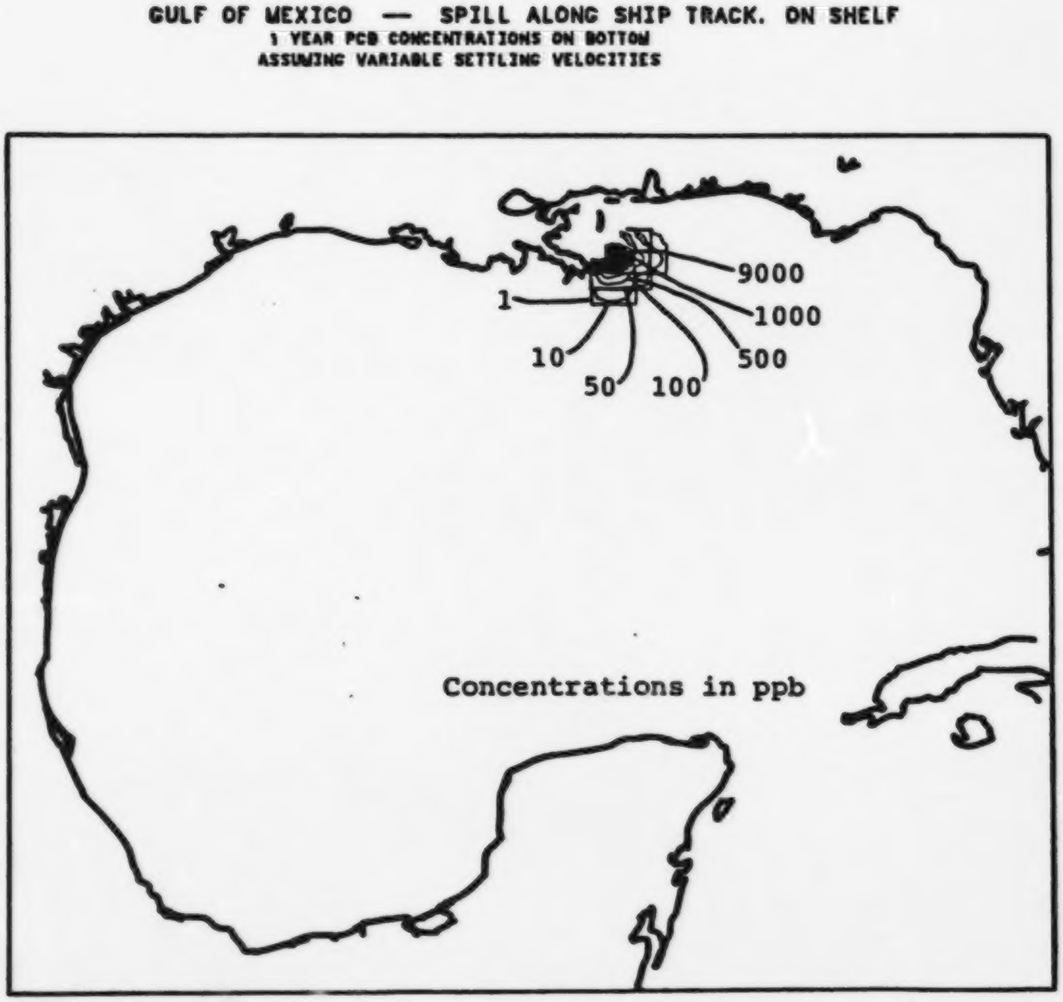
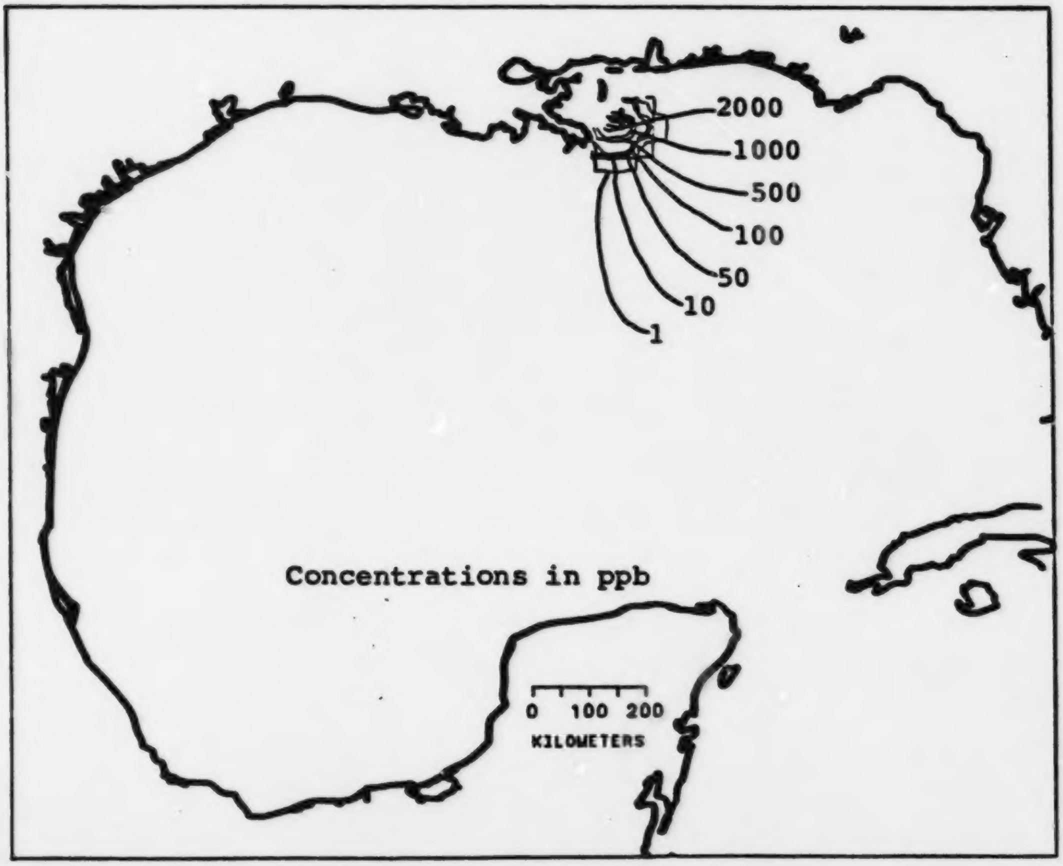

GULF OF MEXICO — SPILL ALONG SHIP TRACK. ON SHELF
1 YEAR PCB CONCENTRATIONS ON BOTTOM
ASSUMING VARIABLE SETTLING VELOCITIES
9000
1000
500
1
10
50
100
Concentrations in ppb

Figure 6- 5a. PCB concentrations on bottom due to sinking spill on shelf, 1 year after spill of entire cargo.



GULF OF MEXICO — SPILL ALONG SHIP TRACK. ON SHELF
1 YEAR PCB CONCENTRATIONS ON BOTTOM
ASSUMING VARIABLE SETTLING VELOCITIES

Figure 6-5b. As 6-5a, except for a spill of ¼ of cargo.

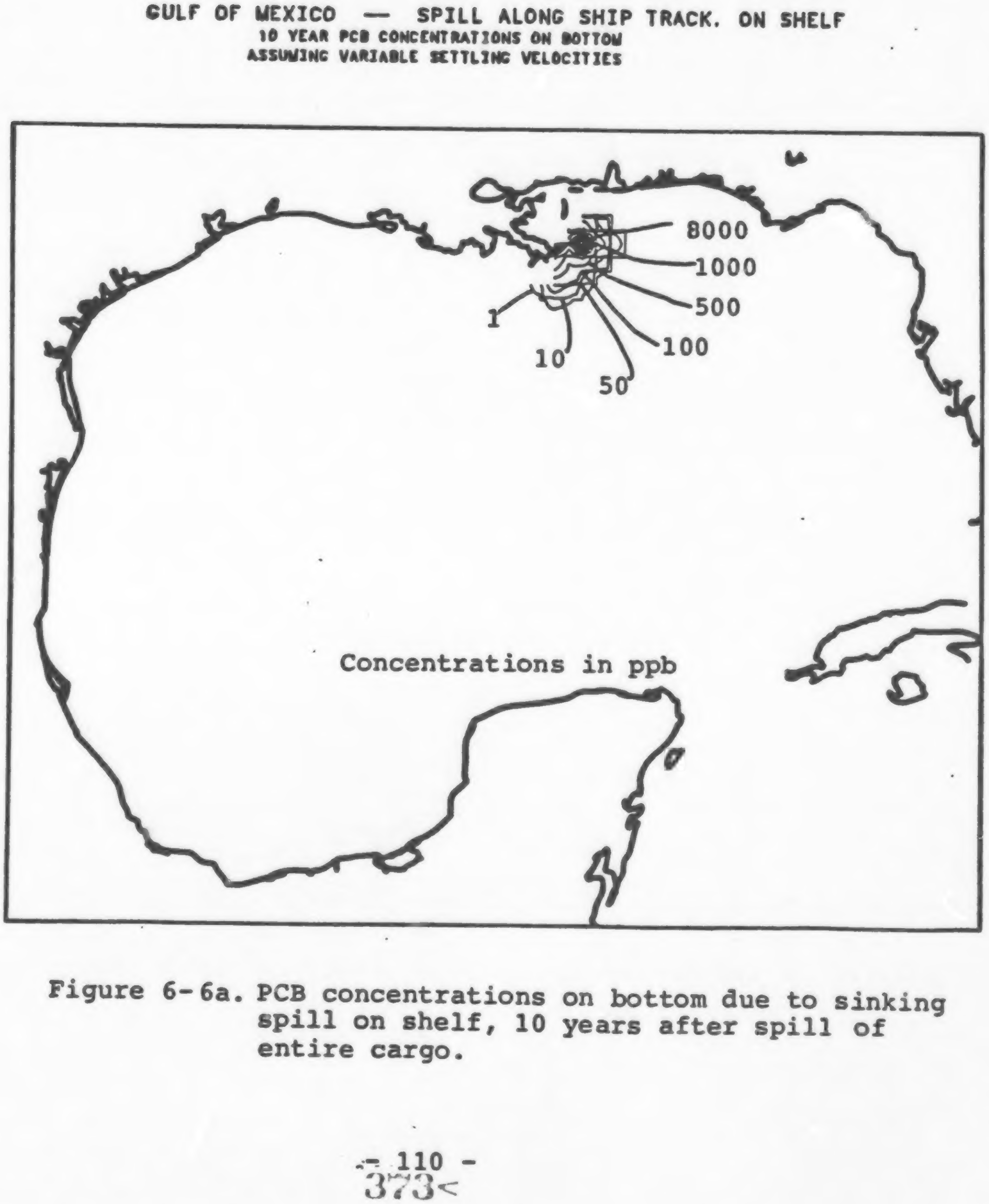
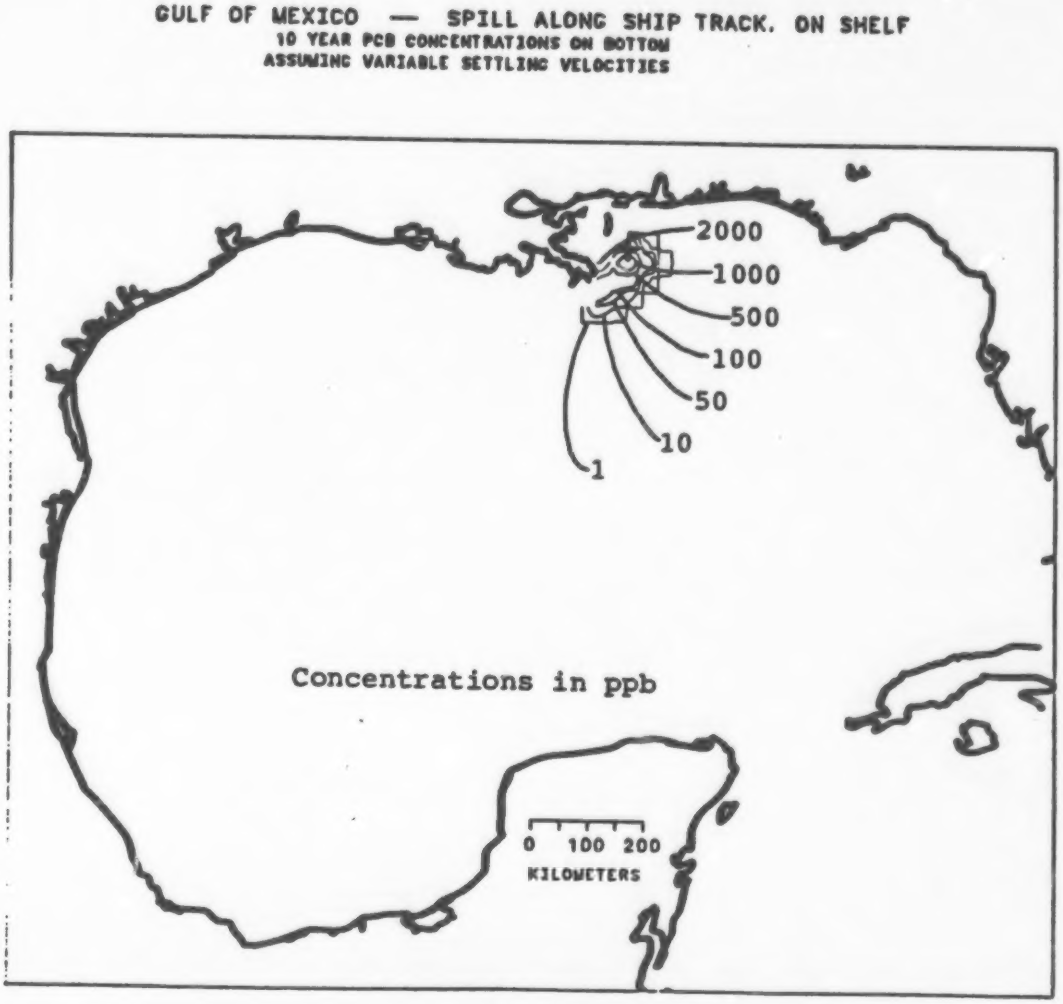

GULF OF MEXICO — SPILL ALONG SHIP TRACK. ON SHELF
10 YEAR PCB CONCENTRATIONS ON BOTTOM
ASSUMING VARIABLE SETTLING VELOCITIES
8000
1000
500
100
50
10
1
Concentrations in ppb

Figure 6-6a. PCB concentrations on bottom due to sinking spill on shelf, 10 years after spill of entire cargo.



GULF OF MEXICO — SPILL ALONG SHIP TRACK, ON SHELF
10 YEAR PCB CONCENTRATIONS ON BOTTOM
ASSUMING VARIABLE SETTLING VELOCITIES

Figure 6-6b. As 6-6a, except for a spill of ½ of cargo.

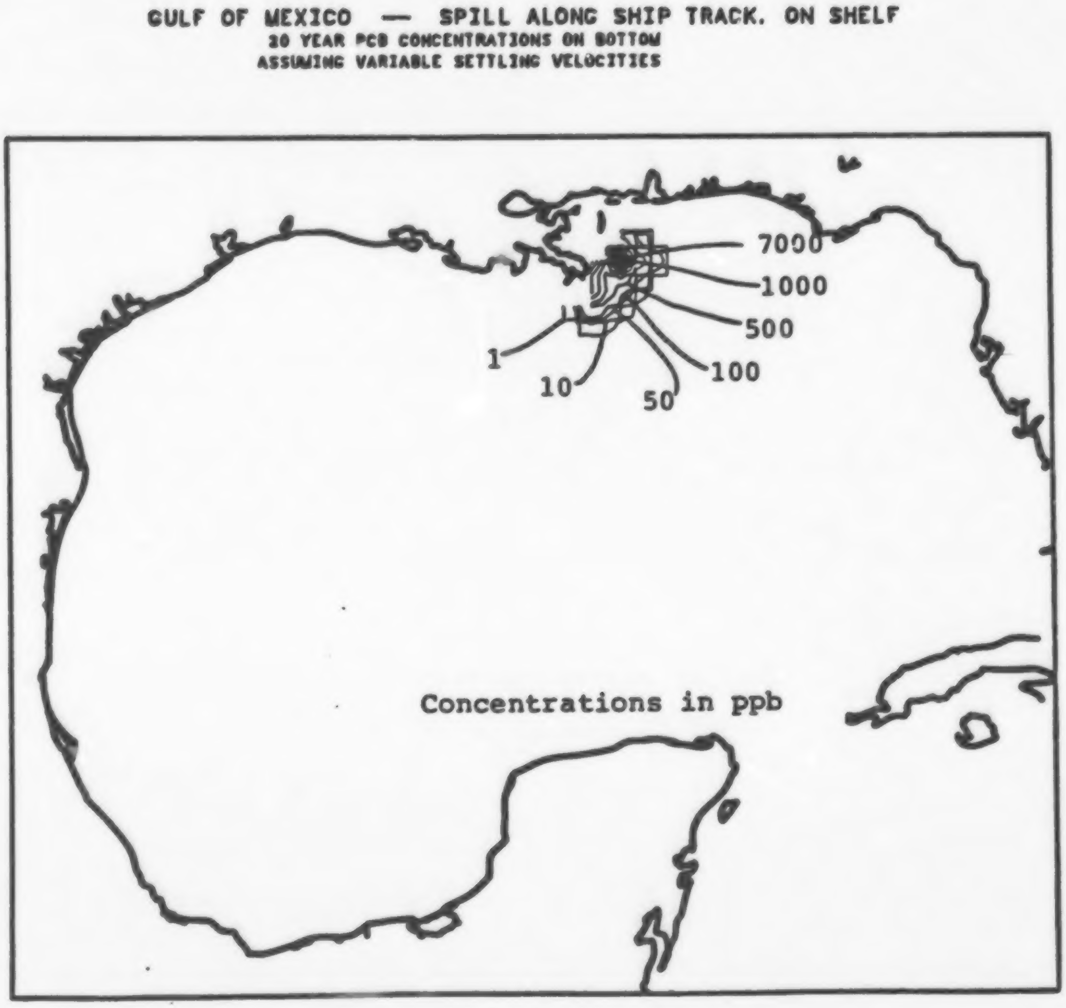

GULF OF MEXICO — SPILL ALONG SHIP TRACK. ON SHELF
20 YEAR PCB CONCENTRATIONS ON BOTTOM
ASSUMING VARIABLE SETTLING VELOCITIES
7000
1000
500
100
50
10
1
Concentrations in ppb

Figure 6-7a. PCB concentrations on bottom due to sinking spill on shelf, 20 years after spill of entire cargo.

GULF OF MEXICO — SPILL ALONG SHIP TRACK. ON SHELF
20 YEAR PCB CONCENTRATIONS ON BOTTOM
ASSUMING VARIABLE SETTLING VELOCITIES

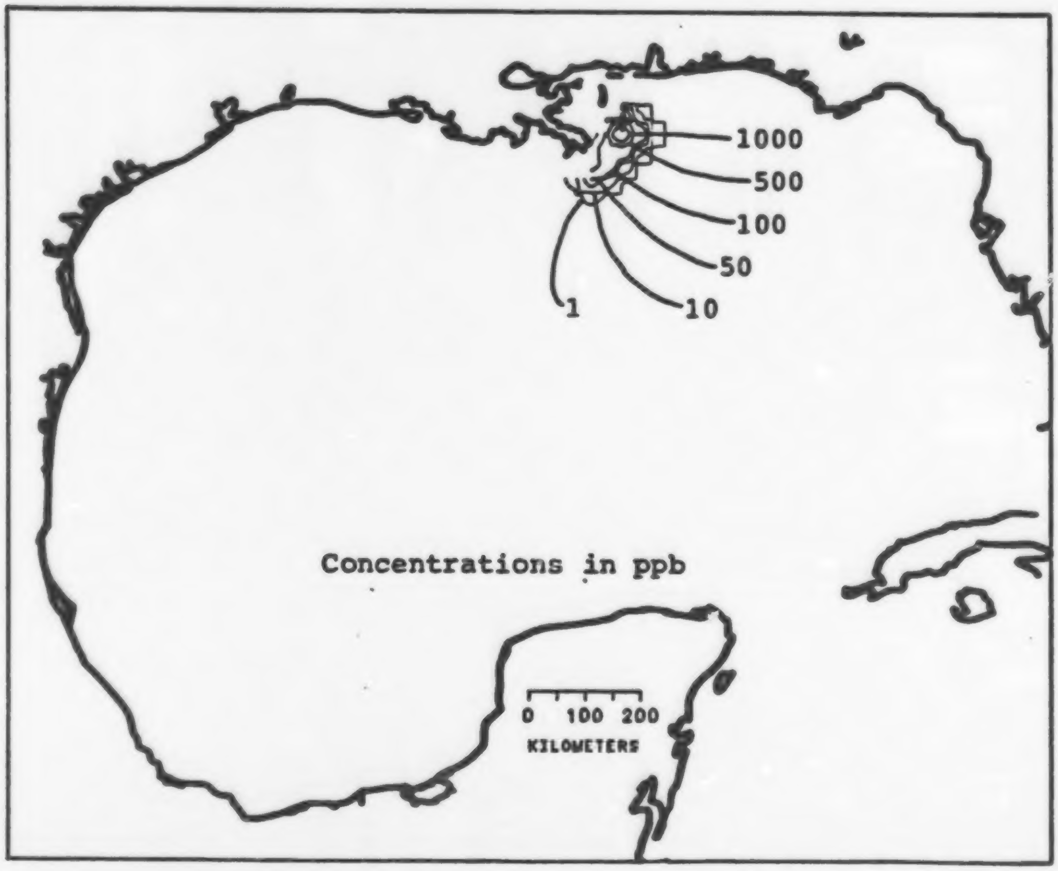


Figure 6-7b. As 6-7a, except for a spill of ¼ of cargo.

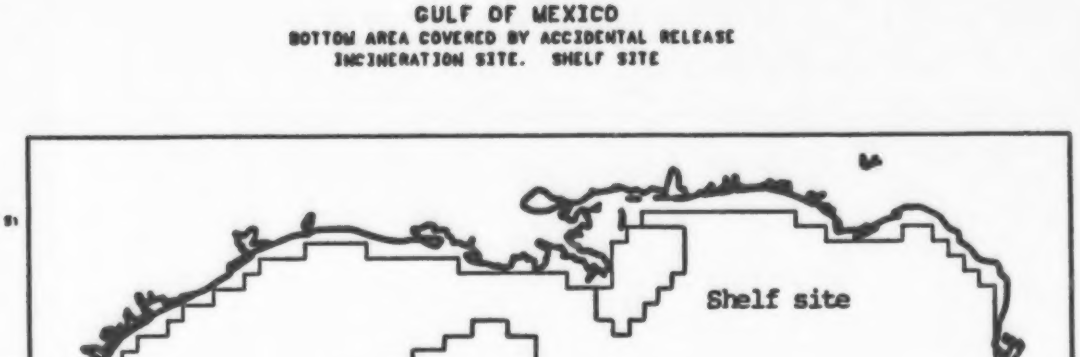

GULF OF MEXICO
BOTTOM AREA COVERED BY ACCIDENTAL RELEASE
INCINERATION SITE. SHELF SITE
Shelf site
Incineration (offshore) site

Figure 6-8. Bottom areas of elevated PCB concentrations used in ecosystem effects model.

by the PCBs and the more volatile medium with which they are mixed. Surface concentrations of PCB averaged over the top 1 m of the water column are shown in Figures 6-9 through 6-12. Some portions of the slick come ashore on the Mississippi Delta, while the remainder of the slick can be seen to move eastward along the coastline due to the surface currents, with maximum concentrations decreasing with time. The actual behavior of a slick in this area will be governed by the specific weather situation at the time of the spill. The transport shown here reflects only an estimate of mean coastal circulation and wind-driven effects, and lacks the detail necessary to estimate the extent of or ecological consequences of coastal impacts in this topographically complex delta region.

Oil spill observations and modeling techniques (Spaulding et al. 1982) suggest that a 4000 MT slick of material similar in consistency to crude oil will spread to a terminal area of about 1.8 km^2 within a day. After this time, the slick will begin to break up into patches, and these will in turn disperse over a wider and wider area. It can be shown (Csanady 1973) that the standard deviation S of the patch distribution is approximately

$$S = (2Dt)^{1/2} \qquad 6.1.3\text{-}1$$

in which D is diffusivity and t is time. If R is the initial slick radius (R = 756 m), and we take D = 100 m^2/sec, then we can apply Equation 6.1.3-1 to estimate areal patch distributions. Thus after 2 days, the ocean surface area over which the patches

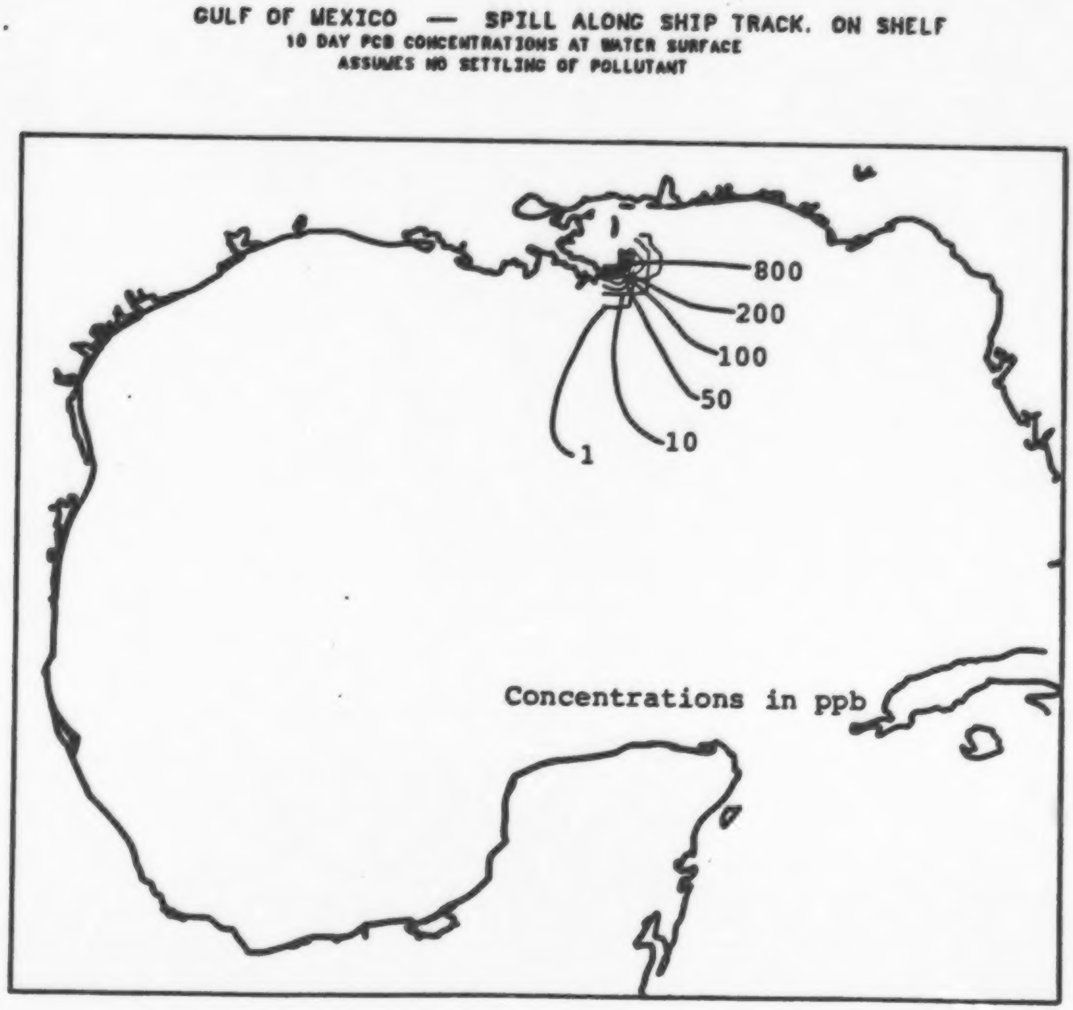

GULF OF MEXICO — SPILL ALONG SHIP TRACK, ON SHELF
10 DAY PCB CONCENTRATIONS AT WATER SURFACE
ASSUMES NO SETTLING OF POLLUTANT
800
200
100
50
10
1
Concentrations in ppb

Figure 6- 9 . PCB concentrations in upper 1 m of water column due to floating spill on shelf, 10 days after spill.

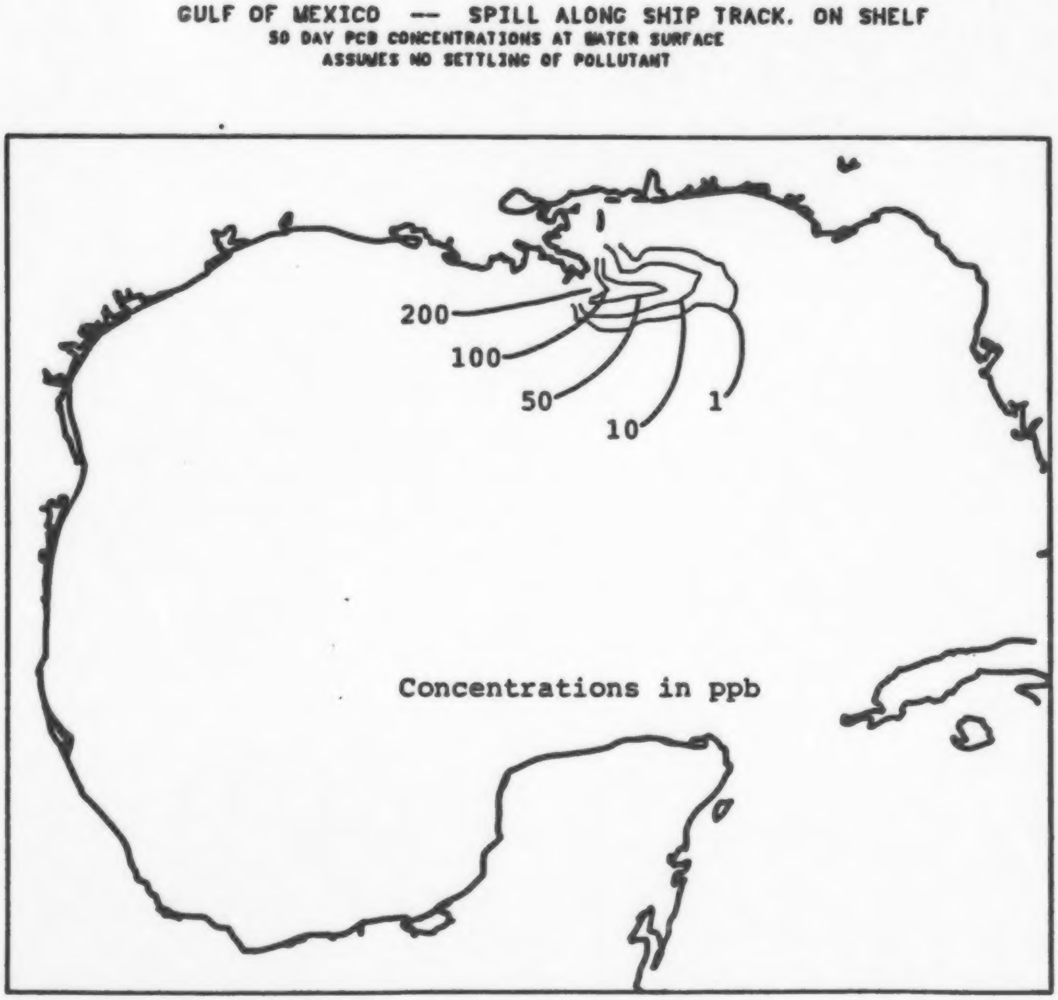
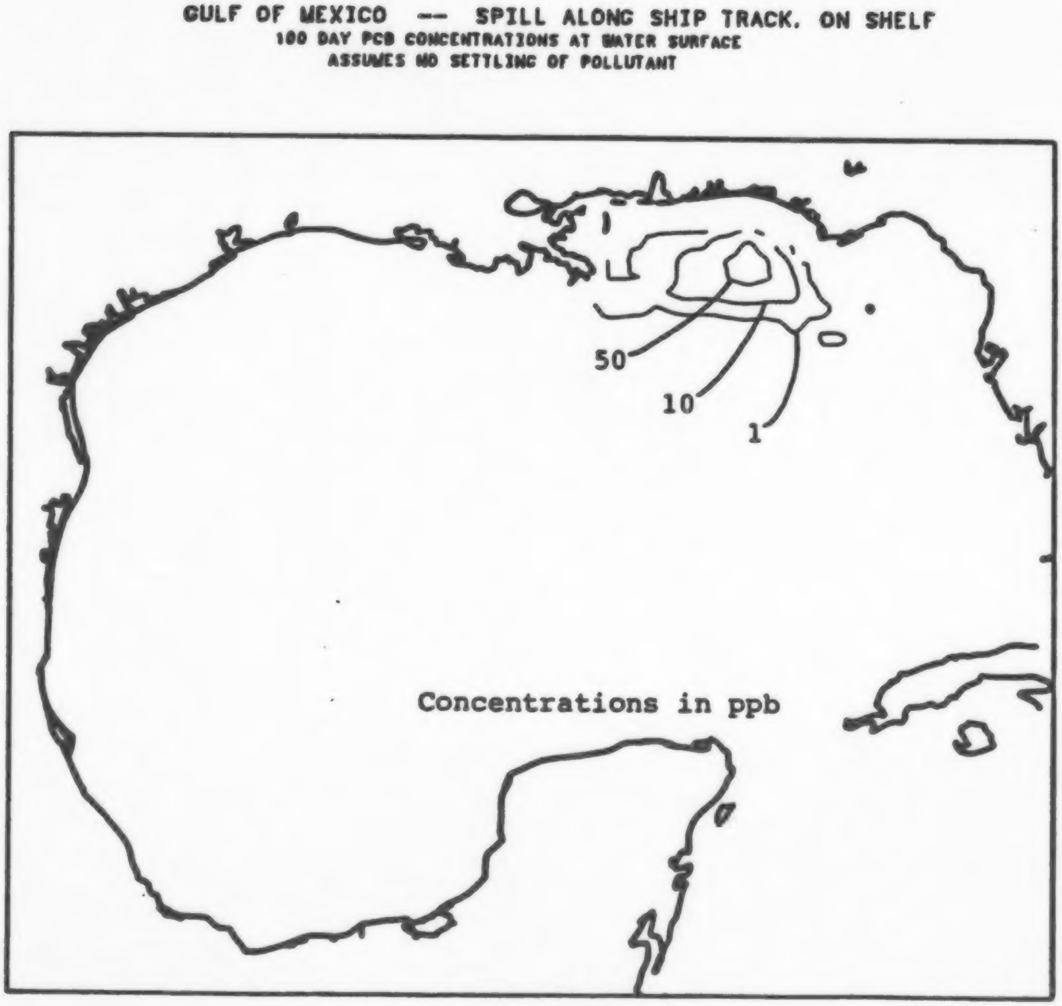

GULF OF MEXICO -- SPILL ALONG SHIP TRACK. ON SHELF
50 DAY PCB CONCENTRATIONS AT WATER SURFACE
ASSUMES NO SETTLING OF POLLUTANT
200
100
50
10
1
Concentrations in ppb

Figure 6-10. PCB concentrations in upper 1 m of water column due to floating spill on shelf, 50 days after spill.



GULF OF MEXICO -- SPILL ALONG SHIP TRACK. ON SHELF
100 DAY PCB CONCENTRATIONS AT WATER SURFACE
ASSUMES NO SETTLING OF POLLUTANT

Figure 6-11. PCB concentrations in upper 1 m of water column due to floating spill on shelf, 100 days after spill.

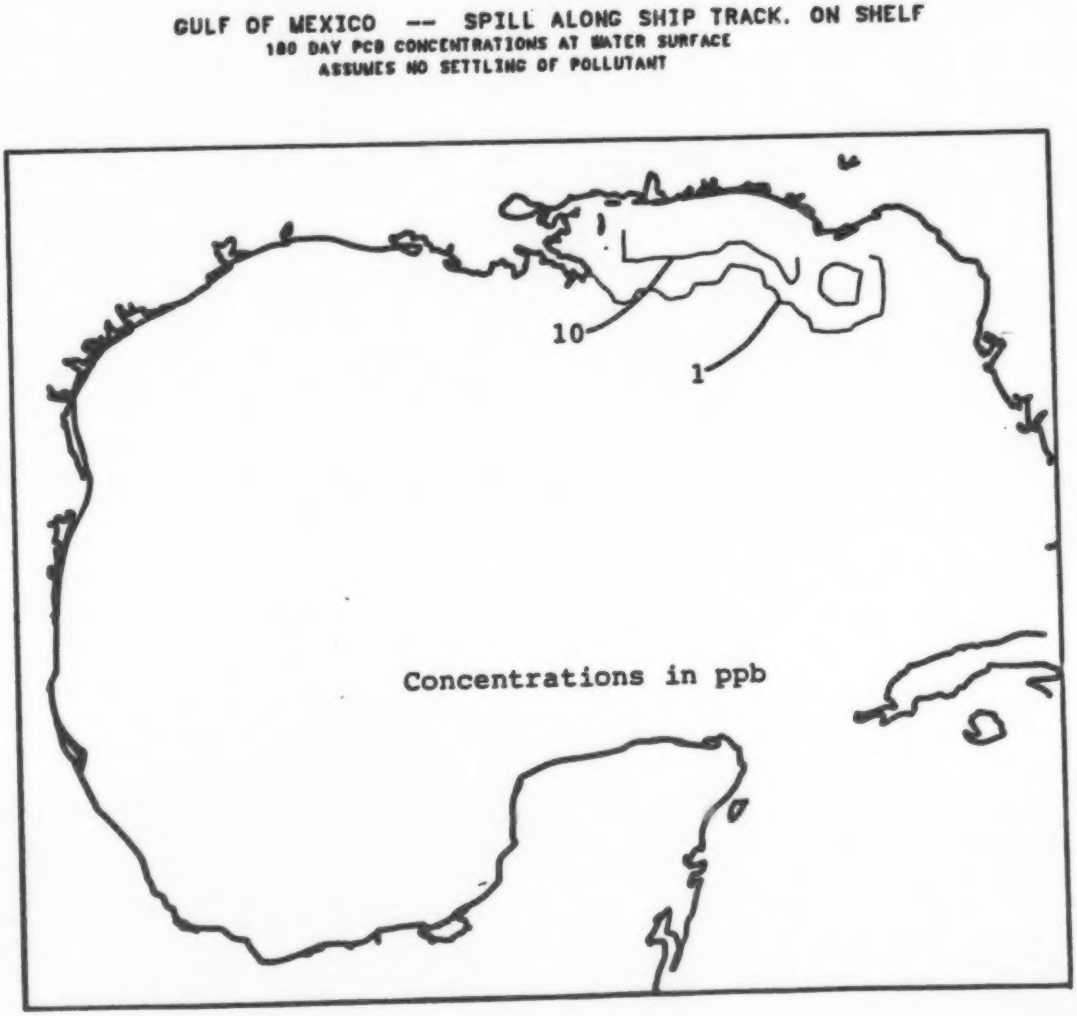

GULF OF MEXICO -- SPILL ALONG SHIP TRACK, ON SHELF
180 DAY PCB CONCENTRATIONS AT WATER SURFACE
ASSUMES NO SETTLING OF POLLUTANT
10
1
Concentrations in ppb

Figure 6-12. PCB concentrations in upper 1 m of water column due to floating spill on shelf, 180 days after spill.

will have dispersed is approximately 109 km^2, and after 10 days, 540 km^2. This estimate assumes random turbulent dispersion. As can be seen from Figure 6-9, the inclusion of horizontal shears in the advective field may considerably increase the areal extent over which such a slick may be dispersed. As discussed in Section 6.3, a slick continuing PCBs will lose mass to the atmosphere and water column as it is advected along the surface. The lighter fractions of the slick will evaporate most rapidly, leaving behind the heavier residual fractions which include the PCBs of concern. The majority of the surface PCB removal will probably be due to dispersion into the water column (JRB 1984) or sinking due to density increase as the lighter waste fractions are removed. At wind speeds of 7 to 8 m/sec and an air temperature of 90° F, reasonable conditions for the Gulf of Mexico, between 40% and 50% of a waste slick will have been list due to weathering within 90 hours of release. About 90% of this loss will be due to dispersion into the water column (JRB 1984). The mass loss rate decreases as a function of time, and very little further loss is expected after about 90 hours. Because weathering loss estimates are not included in the present simulations, and because PCBs are associated with the less volatile fractions of the slick, the surface concentrations shown in Figure 6-11 and 6-12 are probably 30% or 40% too high. It is also quite likely that the PCB components of the slick would sink or settle with suspended particulate matter within a month after a spill.

Response of Shelf Ecosystem to Spills of PCBs

In the case of a spill of PCBs, the bulk of the material will most likely to sink to the bottom since the density of PCBs is greater than that of water. According to the hydrodynamic model results described above, the spill would distribute itself over a bottom area of about 1.75 x 10^{10} m^2. The average sediment concentration in this area, assuming a 1400 MT spill and penetration to 5 cm by wave action and bioturbation, would be about 1 ppm. Water concentrations would quickly (<1 day) return to background levels. The PCBs in the sediments would dissolve (slowly) over time. Figures 6-13 to 6-16 illustrate the effect of this spill on the shelf ecosystem. In Figures 6-13a and 6-14a,b, PCBs are assumed to dissolve and be removed from the sediment at the rate of 1.0 ppb per day. In Figures 6-15a and 6-16a,b, this loss rate is assumed to be 0.1 ppb per day. The two runs represent a range of possible loss rates estimated by the methods of Section 6.1.2, and from previous work with the model system (Reed et al. 1984), although the slower rate is thought to be most realistic.

In both cases, PCB levels in the plankton are not affected by the spill, while levels in the benthos and fish increase by 3-4 orders of magnitude (Figures 6-13, 6-15). PCB levels in demersal and large pelagic fish increase to over 100 ppm. Biomass levels of pelagic organisms are not effected by the spill, but the benthos is severely reduced. Benthic invertebrate predators (#9) are most severely affected in the model simulations, disappearing quickly after the spill. The release

PLOT: BIOM.CONC.S1

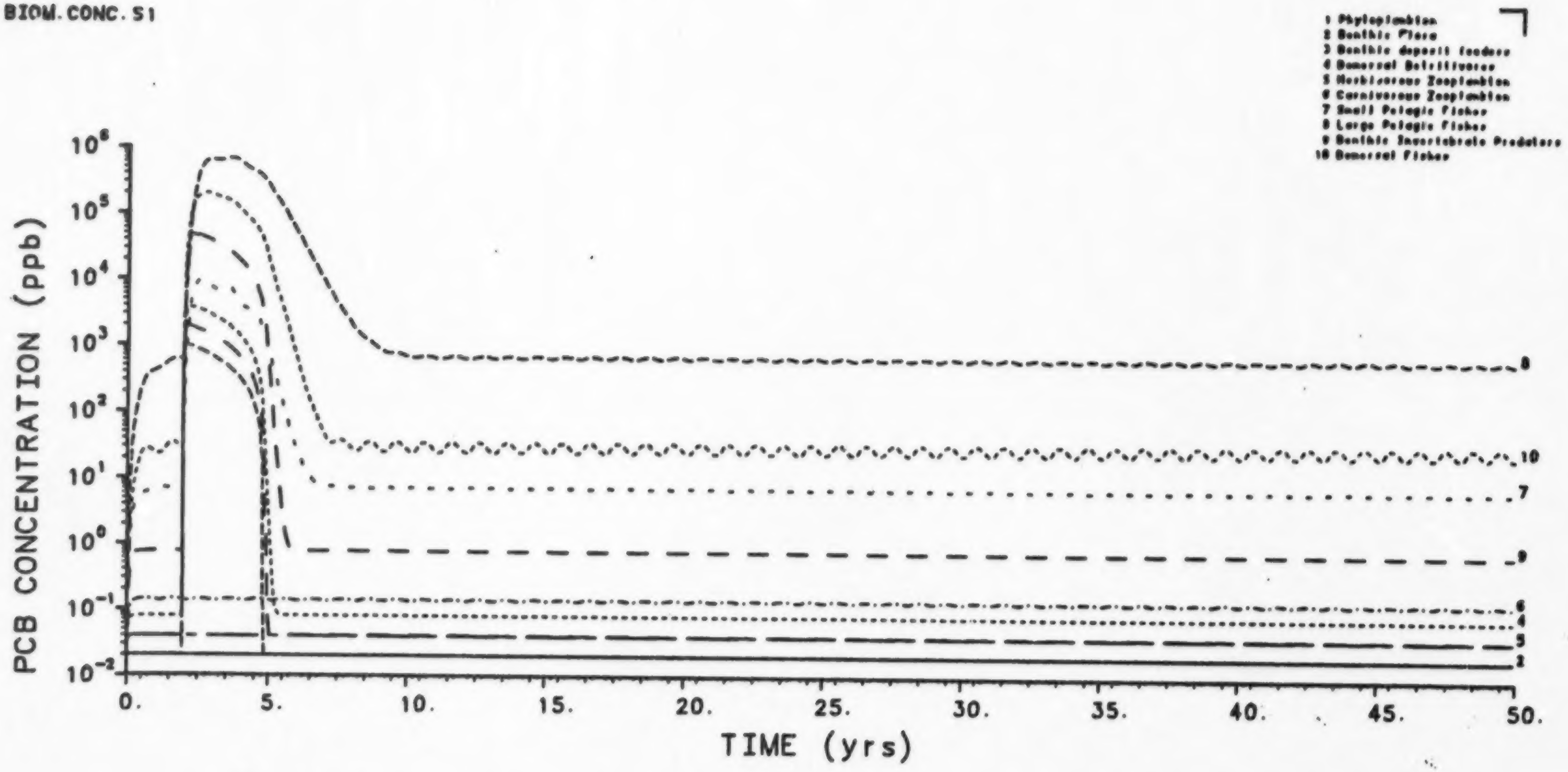


Figure 6- 13a. PCB concentrations in trophic compartments of the shelf ecosystem before and after a spill of the entire cargo occuring at two years. Loss rate from sediment is 1.0 ppb/day.

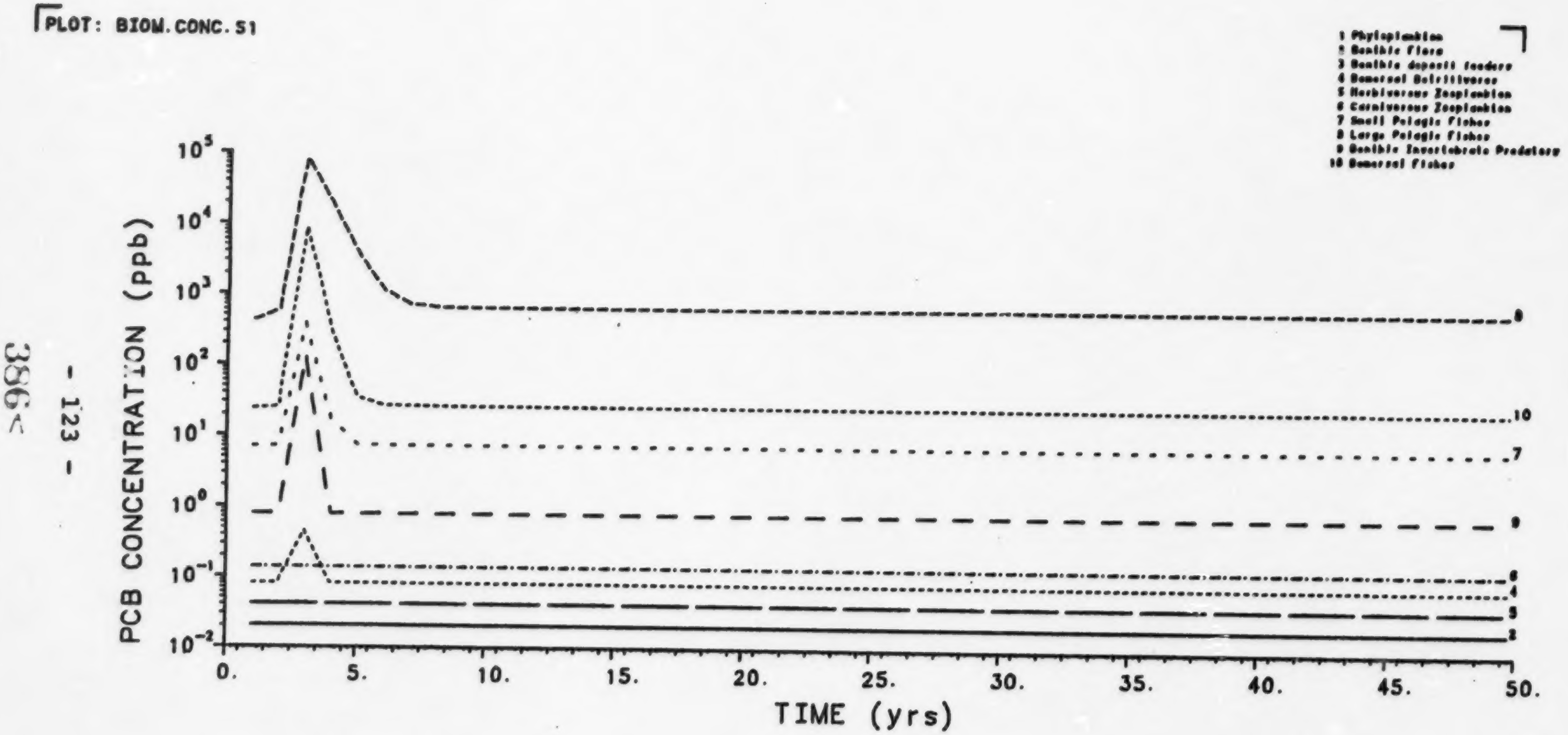


Figure 6-13b. As 6-13a, except 1/4 of the cargo is spilled.

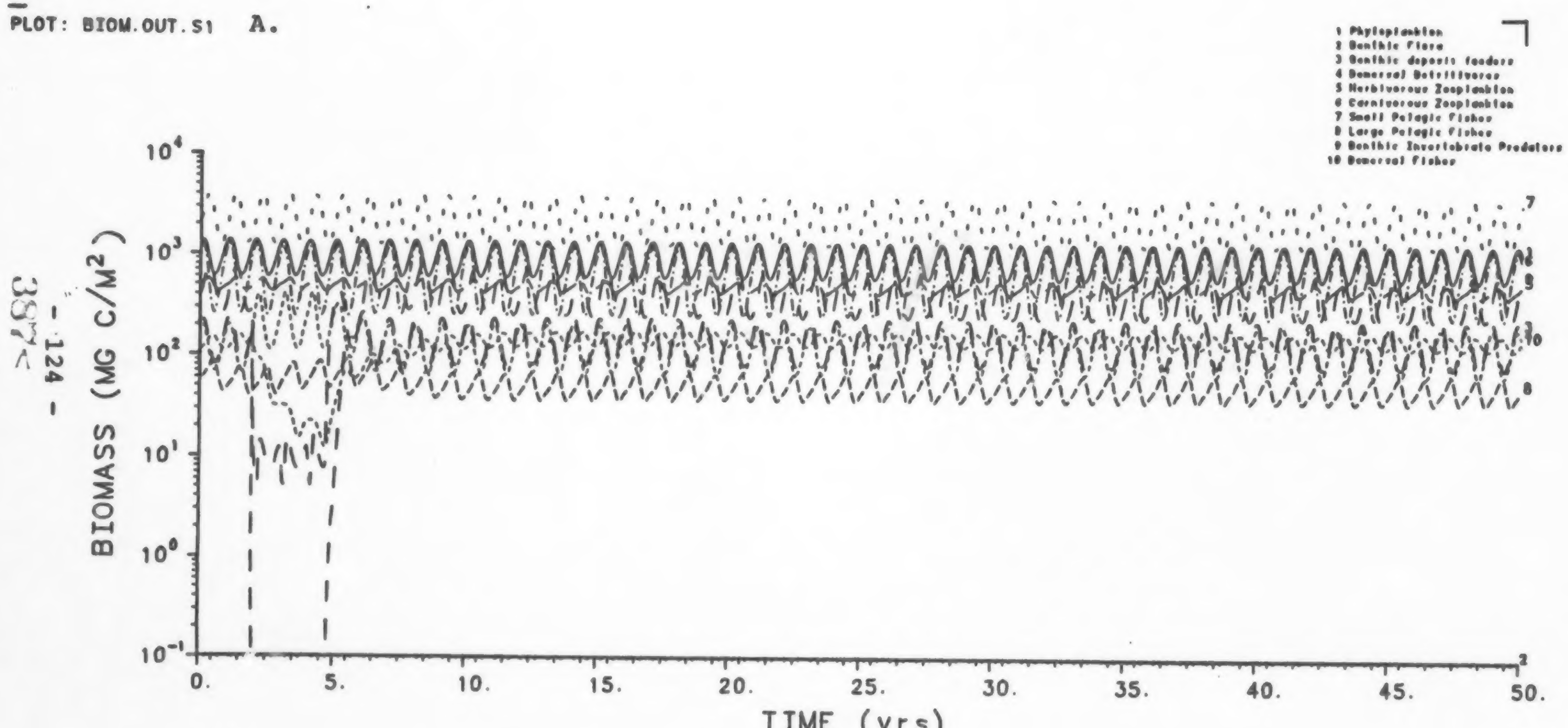


Figure 6-14a. Biomass levels in the shelf ecosystem before and after a spill of the entire cargo of PCB's occurring at two years. Loss rate from sediment is 1.0 ppb/day.

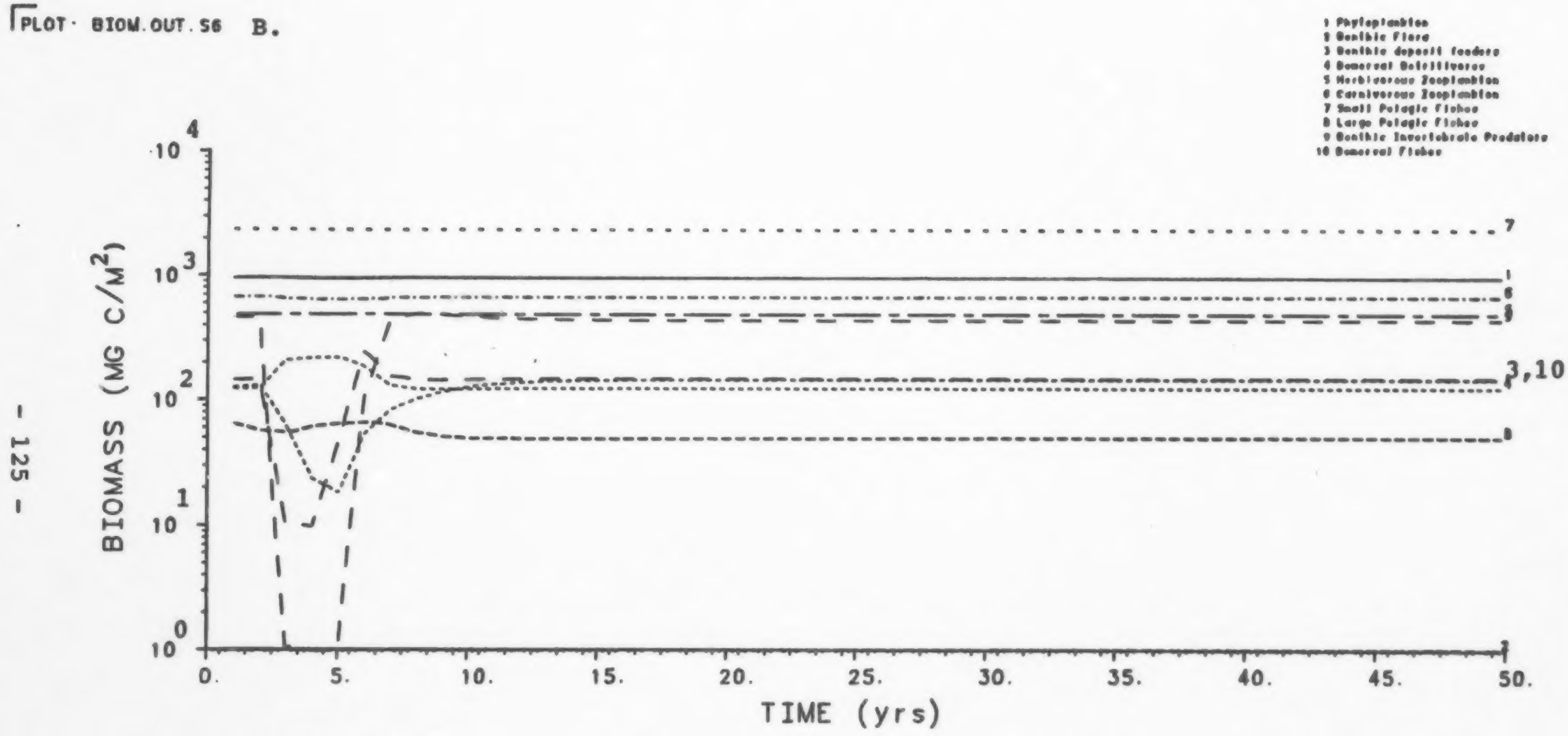


Figure 6-14b. As 6-14a, except yearly average of biomass is plotted versus time.

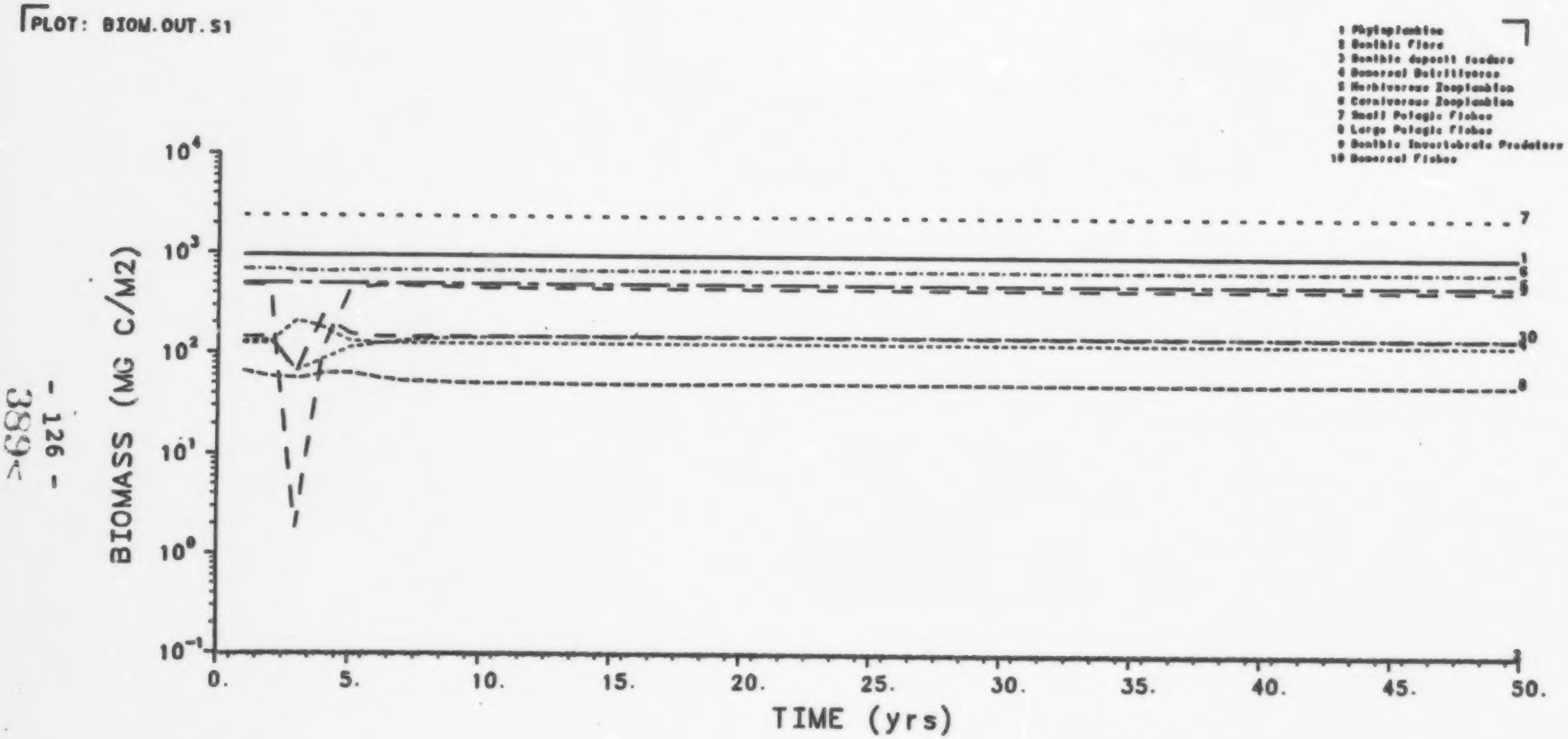


Figure 6-14c. As 6-14b, except 1/4 of the cargo is spilled.

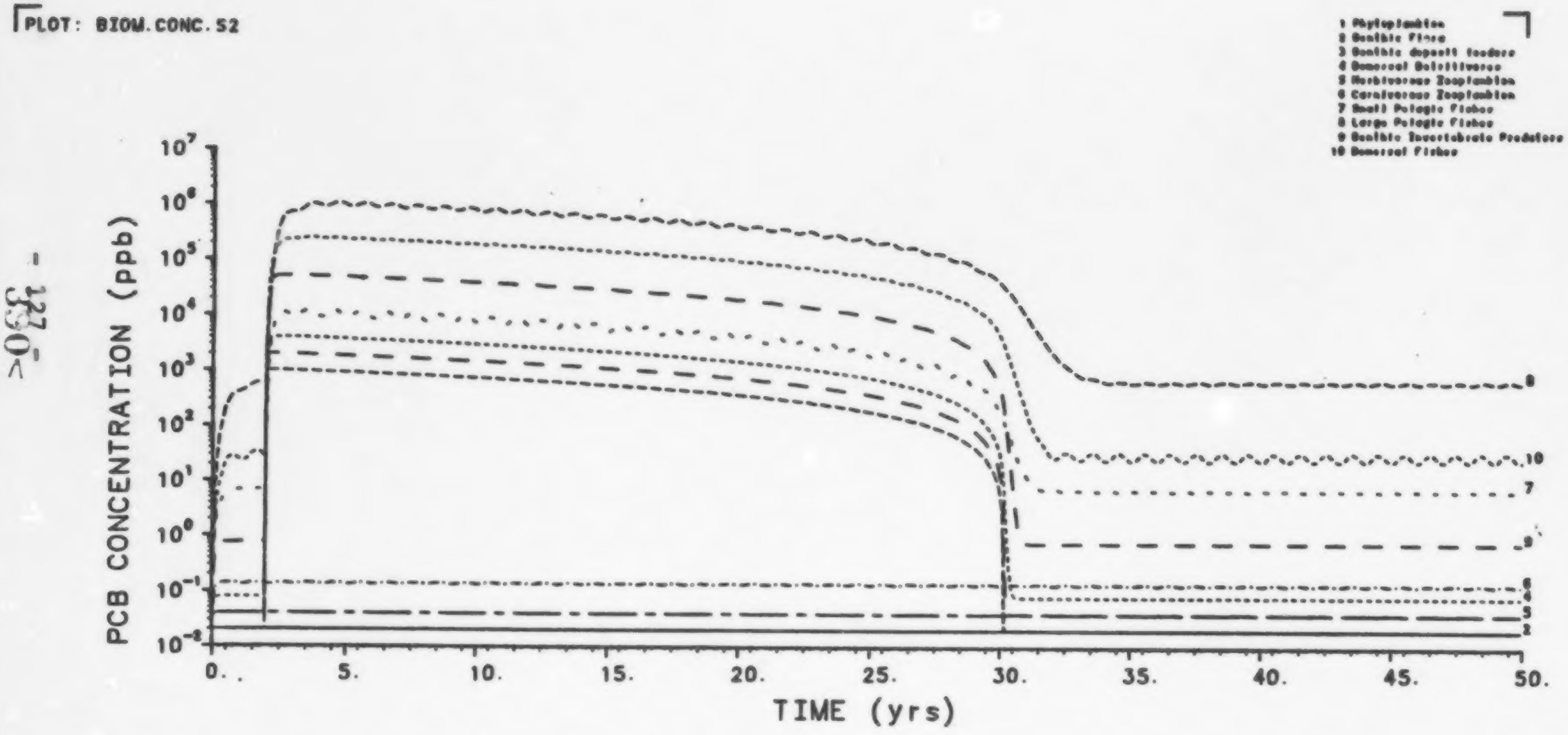


Figure 6-15a. PCB concentrations in trophic compartments of shelf system before and after a spill of the entire cargo occuring at two years. Loss rate from sediment is 0.1 ppb/day.

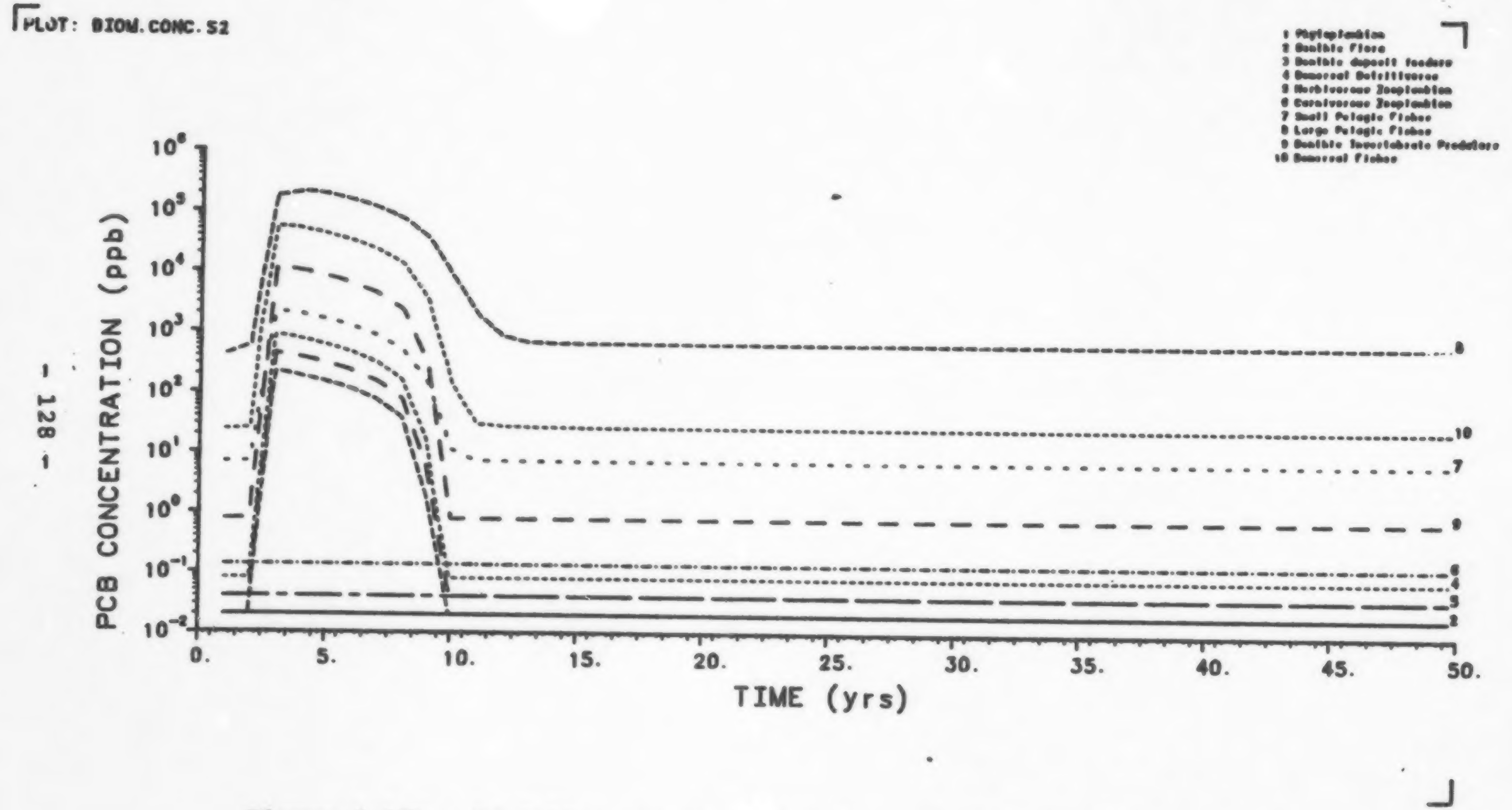


Figure 6-15b. As 6-15a, except 1/4 of the cargo is spilled.

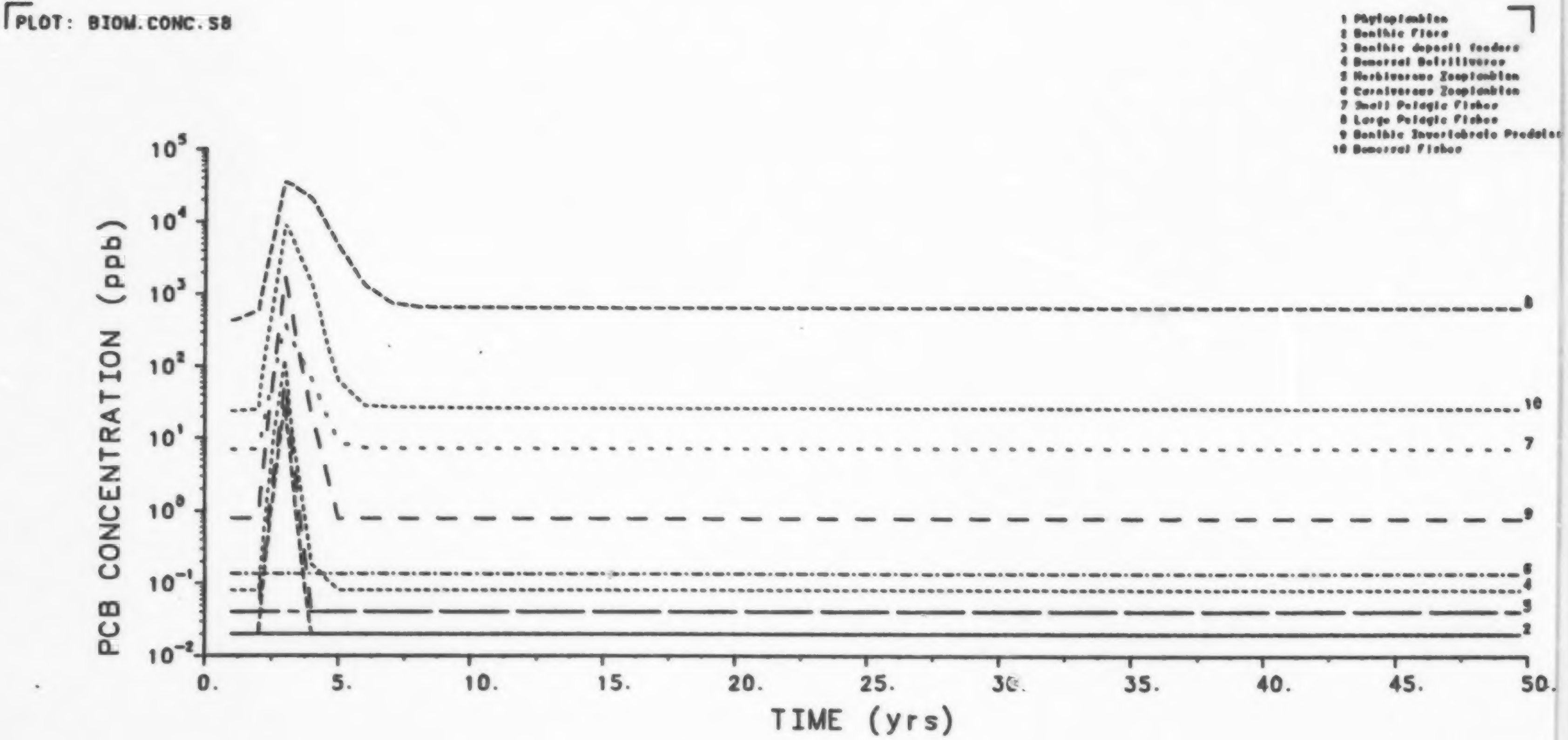


Figure 6-15c. As 6-15a, except 1/16 of the cargo (1/2 tank) is spilled.

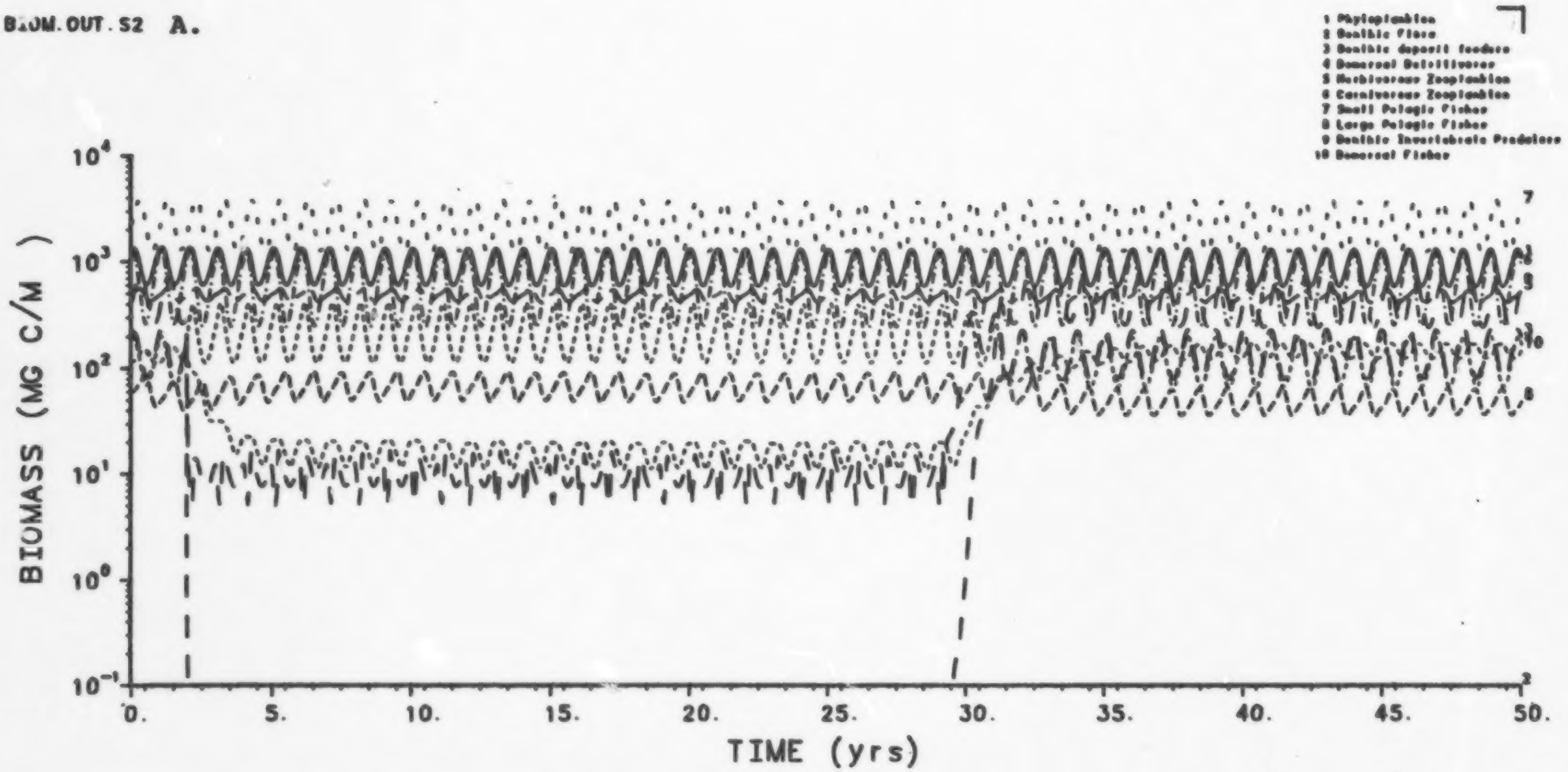


Figure 6-16a. Biomass levels in the shelf ecosystem before and after a spill of the entire cargo of PCB's occurring at two years. Loss rate from sediment is 0.1 ppb/day.

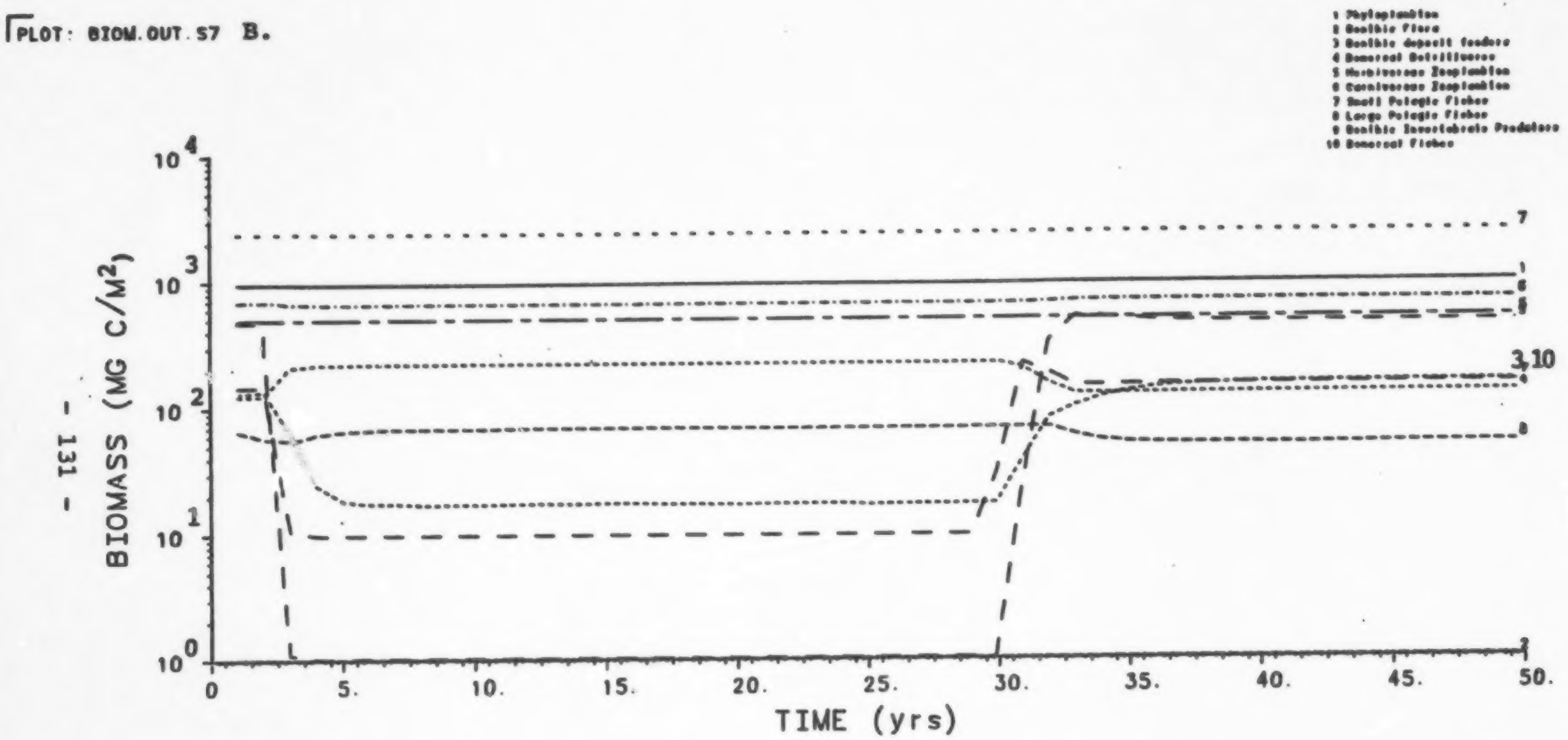


Figure 6-16b. As 6-16a, except yearly average of biomass is plotted versus time.

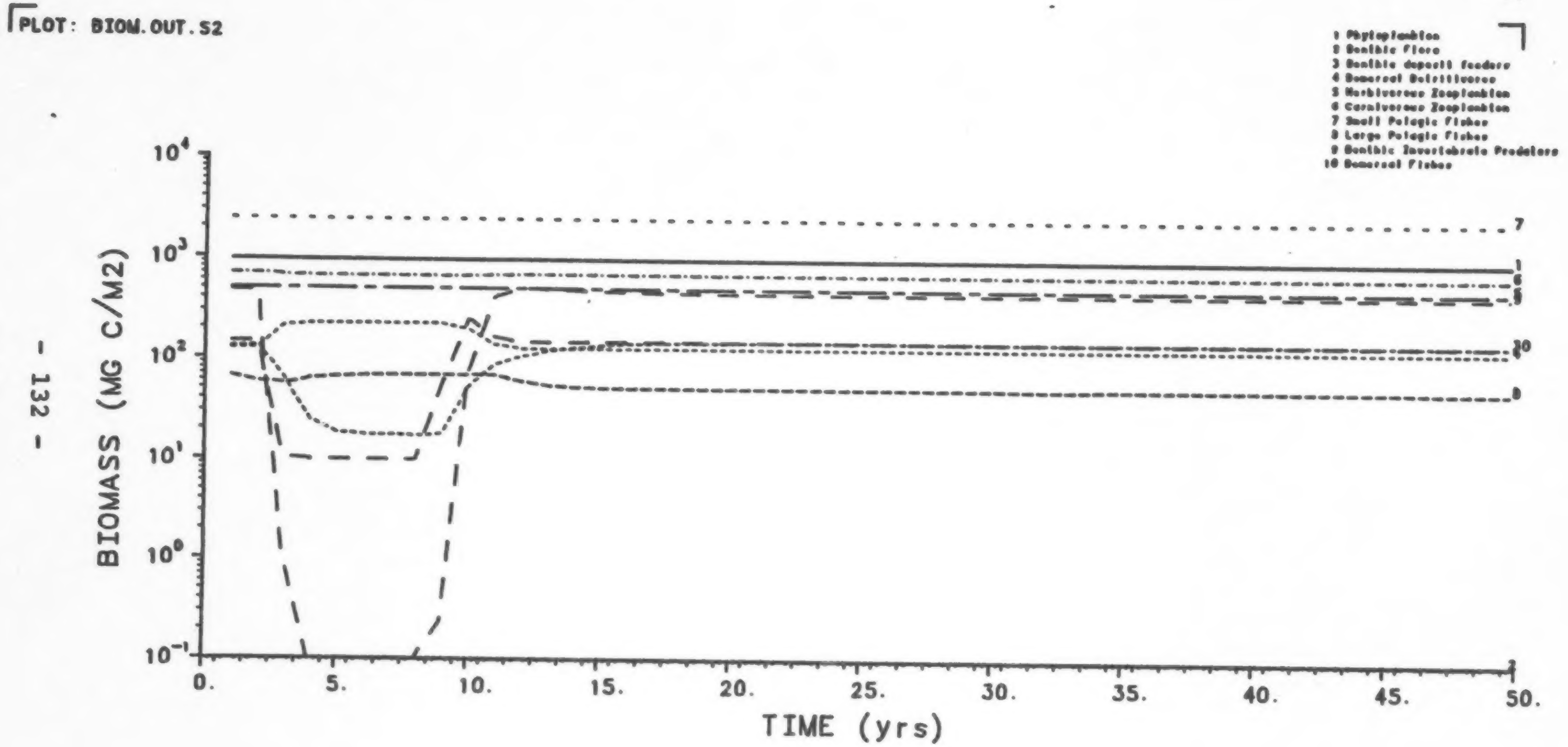


Figure 6-16c. As 6-16b, except 1/4 of the cargo is spilled.

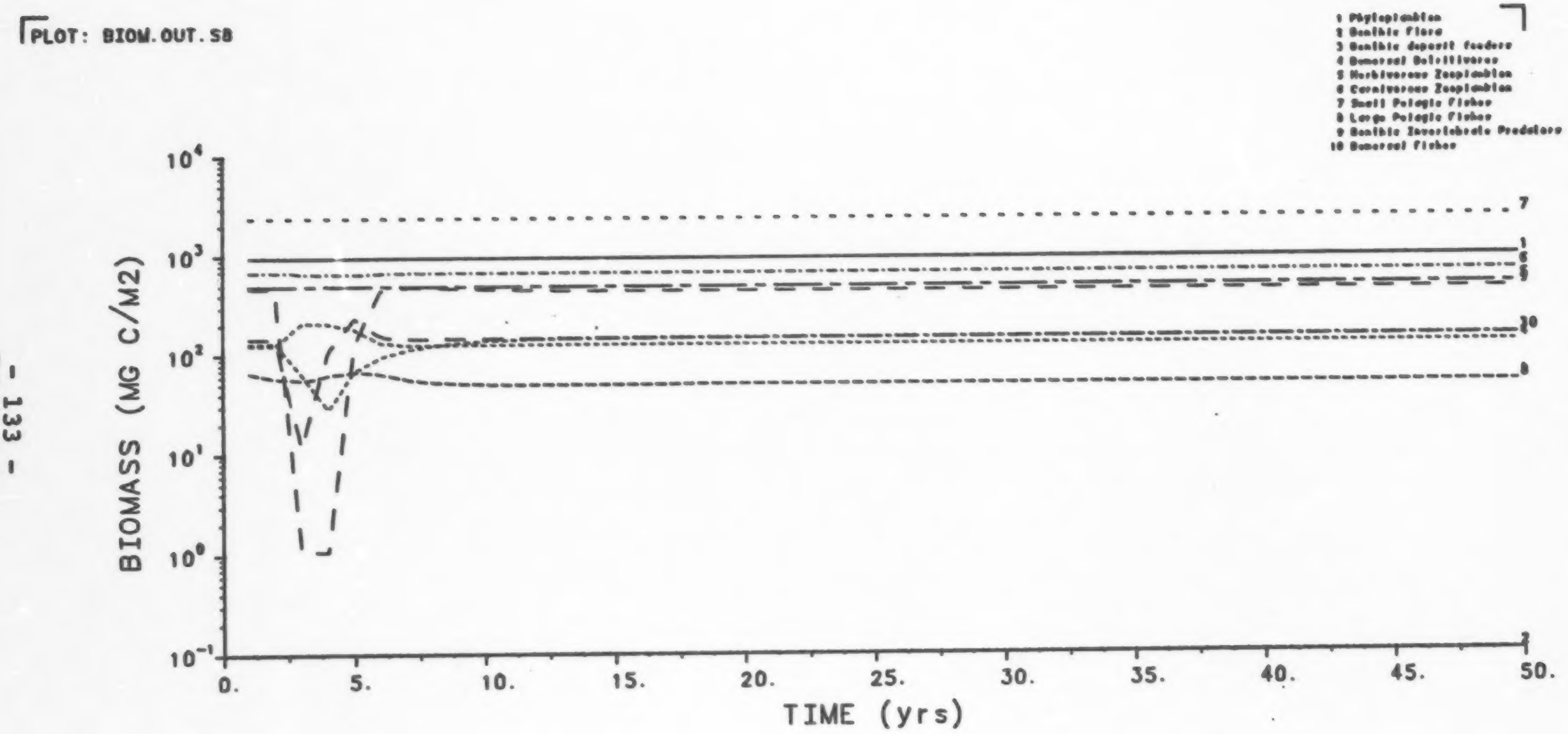


Figure 6-16d. As 6-16b, except 1/16 of the cargo (1/2 a tank) is spilled.

of grazing pressure by compartment #9 on #4, the demersal detritivores (including shrimp), allow compartment #4 to increase slightly after the spill.

Recovery is complete in both cases, but occurs only after nearly all the PCBs have disappeared from the sediment. This is modeled to occur after 4 years when loss is assumed to be 1 ppb per day, and after 30 years when loss is 0.1 ppb per day. However, these rates are quite arbitrary and the likely duration of effects is highly uncertain.

If only two of the eight storage tanks on the ship are lost, the affected seafloor area will be approximately the same. The concentration in the sediments would be approximately 1/4 the value for the entire cargo or 257 ppb. The associated ecosystem response for the same assumed loss rates of 1 ppb per day and 0.1 ppb per day is shown in Figures 6-13b, 6-14c, 6-15b, and 6-16c. In both cases, levels of contamination reach peaks which are an order of magnitude lower than for the full cargo, but the pattern of response is the same. Recovery to pre-spill PCB concentrations occurs after about 1/4 of the time required for recovery after a full-cargo spill (Figures 6-13, 6-15). The Changes in biomass following a spill of 1/4 of the cargo are similar in pattern to those following a full-cargo spill, except the duration of the effect of 1/4 as long.

The intensity of the response is the same if the contamination remains in the sediments for more than 1 year. As described for the entire cargo case above, the benthos, and particularly compartment #9, suffers the highest mortality.

If only 1/2 a tank (1/16 of the entire cargo) is spilled, the average concentration of PCBs in the sediments in the affected area would be 64 ppb. The affected seafloor area would be approximately the same as the 2 and 8 tank spill cases. The ecosystem response for an assumed loss rate of oil ppb per day is shown in Figures 6-15c and 6-16d. The level of contamination of the biota would be an order of magnitude less than that after the 2 tank spill, again with the same pattern of response. Biomass levels of the benthos (compartments #3 and #9) will be depressed initially due to increase mortality, but the system will cleanse itself rapidly enough for recovery within a few years (Figure 6-16d). The pelagic system would not be affected by the spill. Demersal fish will be depressed in numbers for a short while due to decreased food levels in the contaminated area. Demersal detritivores (shrimp) increase slightly in biomass due to release in oredation pressure, but quickly return to pre-spill levels once the sediments are cleansed.

If the PCBs are retained in material which floats as a slick at the surface, the effects will be largely on the surface microlayer and top few meters of the water column, and the associated organisms. PCB levels will be high enough to be lethal to all organisms in the microlayer. Thus the effect on the entire ecosystem is measured by the importance of the microlayer and upper water column to the system, which is unquantified to date. If higher trophic levels obtain significant amounts of food from microlayer-associated organisms, the effect of a slick would be to reduce food resources while the slick is in the area. Once the slick has passed, it is likely that the system will fully recover with minimal contamination. Thus a PCB spill which floats on shelf and offshore areas will most likely have only temporary adverse effects, although these effects could be significant in the short term. Quantitative modeling of this type of pollutant impact is feasible (e.g. Reed 1980, Spaulding et al. 1982), but time has not permitted such model applications in the present study. Contamination could be significant if PCB-contaminated detritus (dead organisms) enters the detrital food chain (i.e. the benthos), but is also likely to be temporary (due to depuration) and diluted quickly.

A PCB spill coming ashore will result in contaminated shoreline. For low energy wetland areas, this contamination will probably be long term and significant, whereas high energy rocky coastlines tend to be rapidly self cleansing. Since most Gulf of Mexico coastal areas are of the former type, coastal impacts of such a spill will very likely be significant.

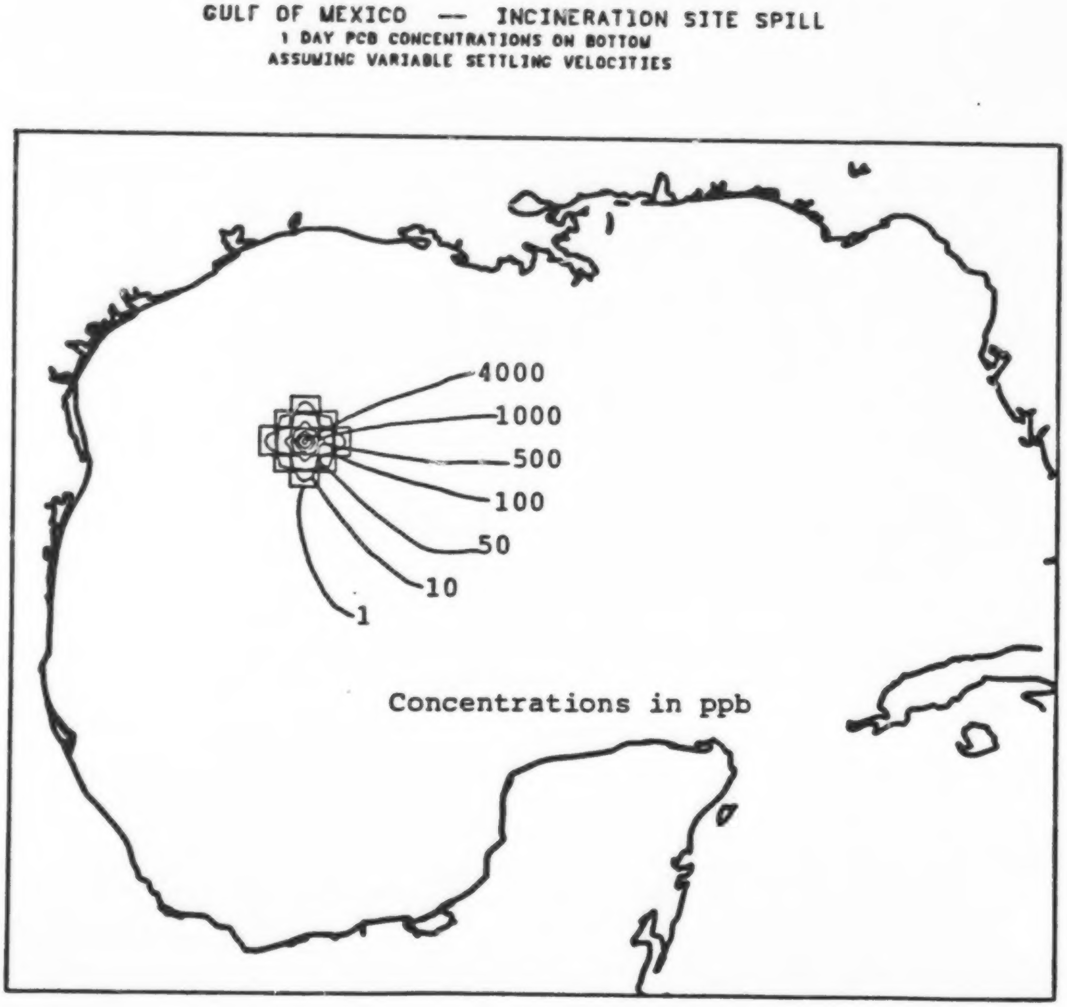
6.1.4 Burn Zone Spill Scenarios

Pollutant Transport Model Predictions

The two scenarios described in Section 6.1.3 were repeated, changing only the spill location. For these simulations, spills are assumed to occur at the center of the incineration site, at 26°40 minutes N, 93° 40 minutes W. Again the entire cargo containing 1400 metric tons of PCBs or two tanks (1/4 cargo) containing 350 mt of PCBs is released instantaneously at the ocean surface. The model surface layer is the upper 60 m of the water column.

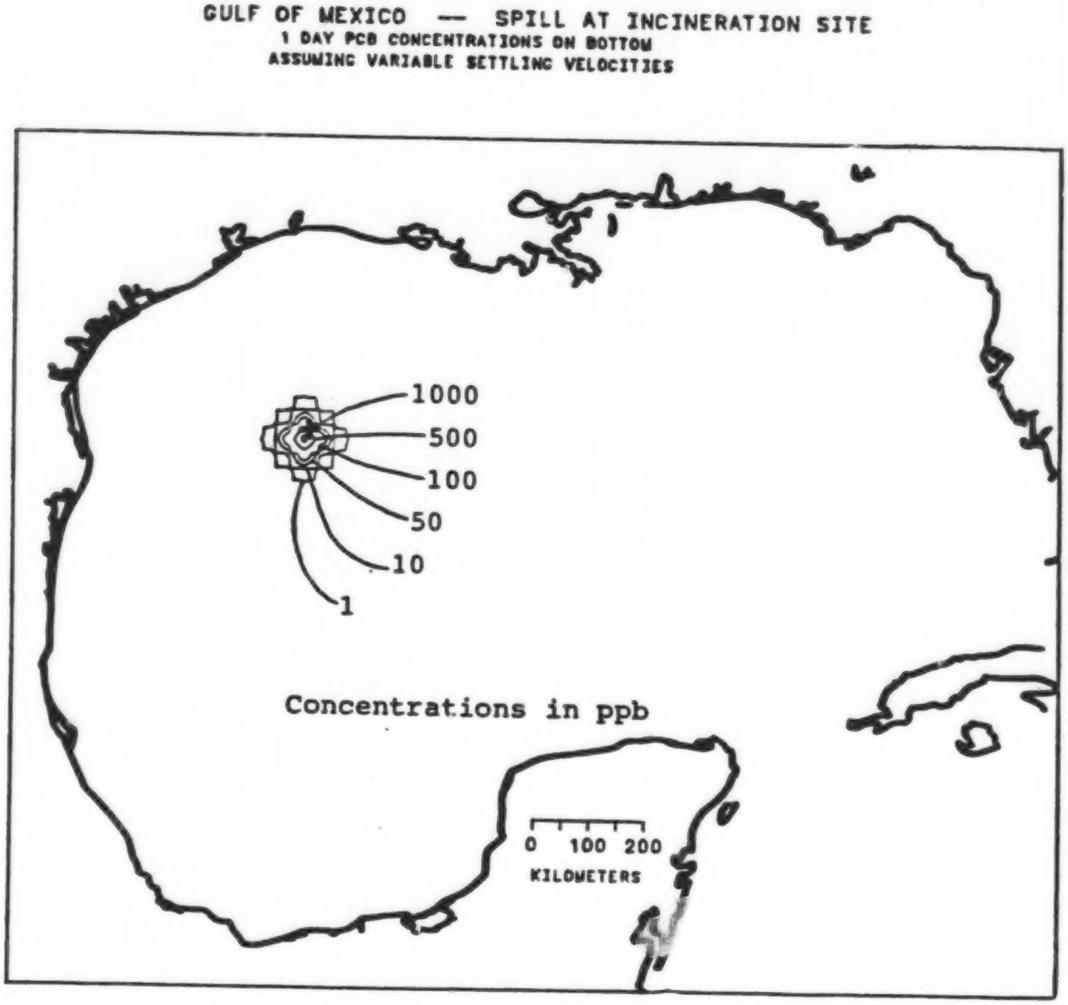
The first scenario, in which all the PCB sinks, uniformly distributes the three settling velocities used previously (0.1, 0.01 and 0.001 m/sec) among the PCBs released. The one-third of the pollutant which settles most rapidly reaches the bottom in less than one day (Figure 6-17). The pollutant associated with the intermediate settling velocity reaches the ocean floor in two to three days' time. The slightly elongated bottom concentration contours (Figure 6-18) show the influence of horizontal advection as the particles descend. This effect is most pronounced in the bottom concentrations shown in Figure 6-19 at 25 days after the spill. By this time, all the particles associated with the slowest settling velocities have reached the bottom. The north-eastward orientation of the contaminated bottom area results from the advection of PCB in this direction by the subsurface current field. Due to the depths to which the PCB settles (generally greater than 100 m), resuspension as a result of storm-generated waves is negligible. The bottom area affected by elevated concentrations of PCB which was used by the the ecosystem effects model is shown in Figure 6-8. As for the continental shelf case, a smaller spill of 1/4 of the total load will cover approximately the same area as the full-cargo spill but at about 1/4 the concentration.

In the second scenario (all the pollutant floats) only the surface currents advect the spilled PCBs. Under this scenario the spill moves generally westward and hits land within 50 days



GULF OF MEXICO -- INCINERATION SITE SPILL
1 DAY PCB CONCENTRATIONS ON BOTTOM
ASSUMING VARIABLE SETTLING VELOCITIES

Figure 6-17a. PCB concentrations on bottom due to sinking spill at incineration site, 1 day after spill of entire cargo.



GULF OF MEXICO — SPILL AT INCINERATION SITE
1 DAY PCB CONCENTRATIONS ON BOTTOM
ASSUMING VARIABLE SETTLING VELOCITIES

Figure 6-17b. As 6-17a, except for a spill of 1/4 of cargo.

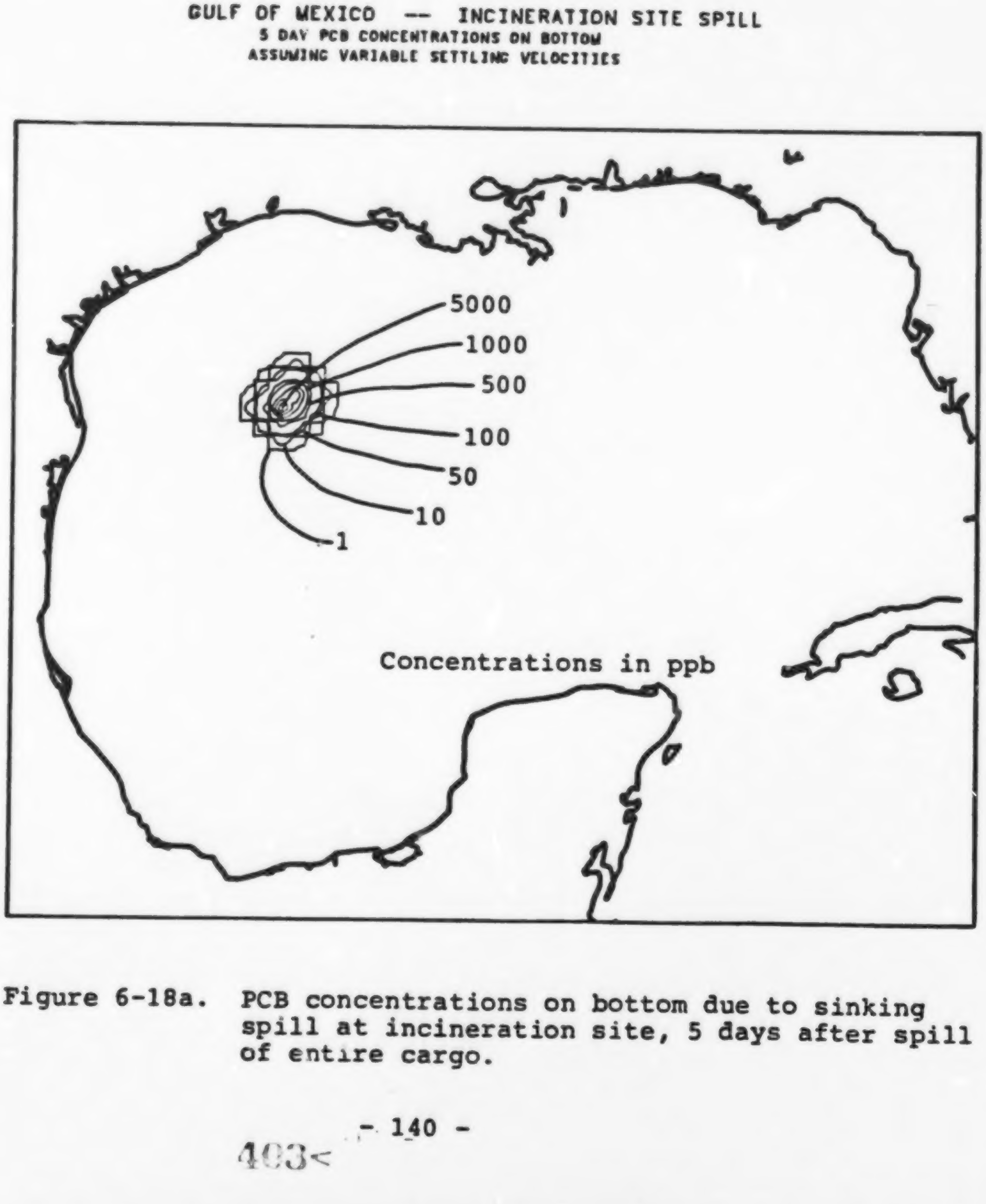
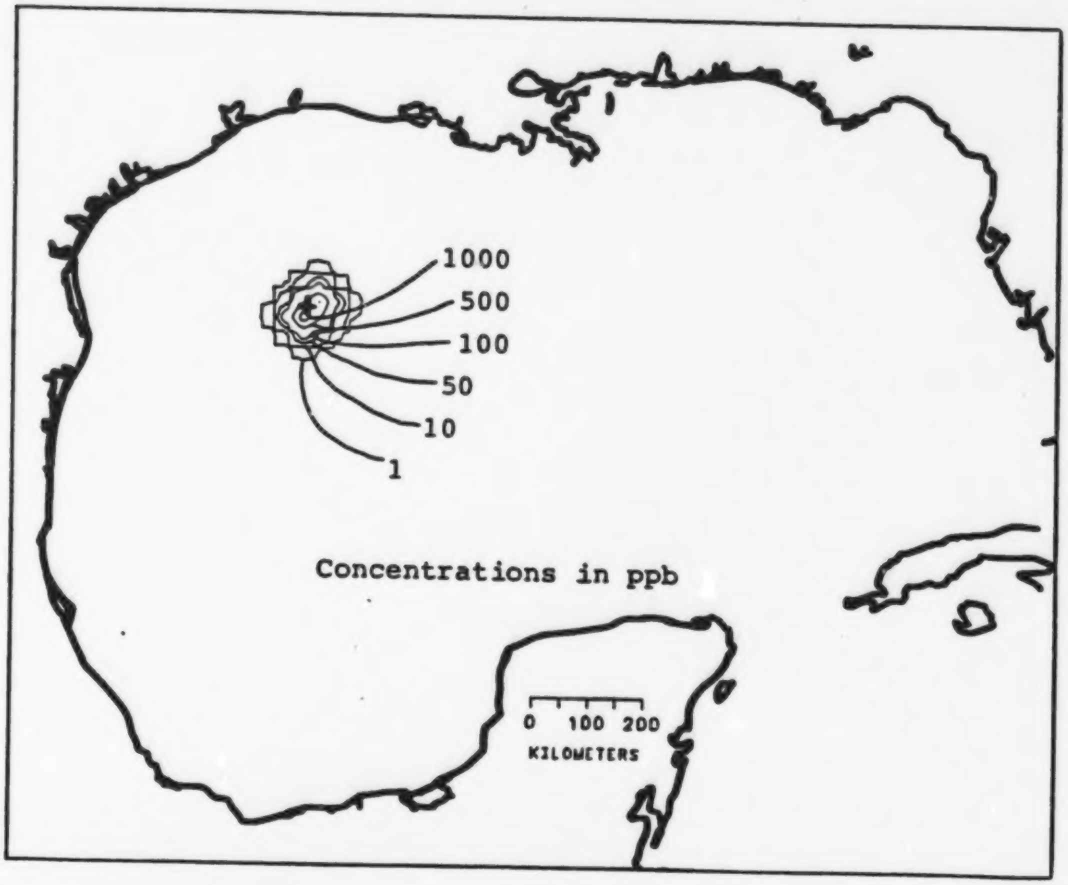

GULF OF MEXICO -- INCINERATION SITE SPILL
5 DAY PCB CONCENTRATIONS ON BOTTOM
ASSUMING VARIABLE SETTLING VELOCITIES
5000
1000
500
100
50
10
1
Concentrations in ppb

Figure 6-18a. PCB concentrations on bottom due to sinking spill at incineration site, 5 days after spill of entire cargo.



GULF OF MEXICO -- SPILL AT INCINERATION SITE
5 DAY PCB CONCENTRATIONS ON BOTTOM
ASSUMING VARIABLE SETTLING VELOCITIES

Figure 6-18b. As 6-18a, except for a spill of 1/4 of cargo.

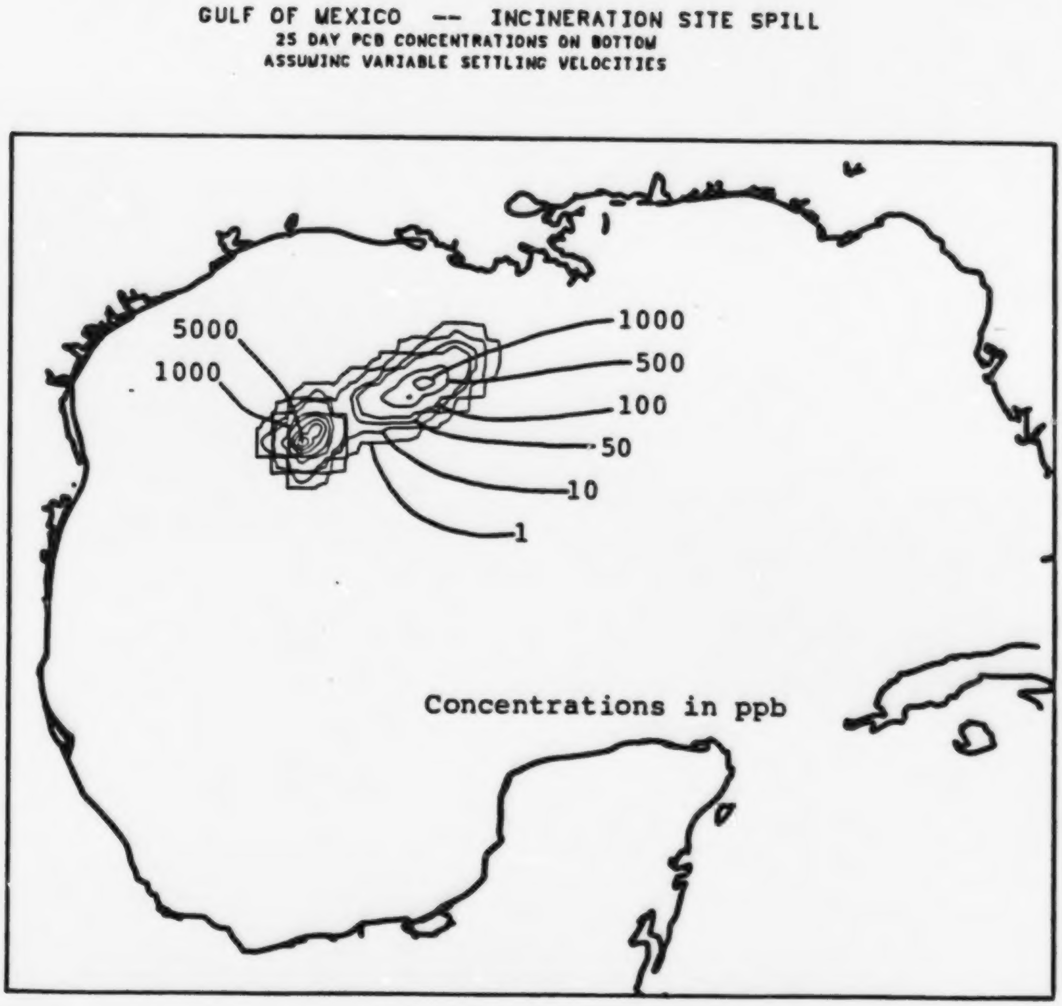

GULF OF MEXICO -- INCINERATION SITE SPILL
25 DAY PCB CONCENTRATIONS ON BOTTOM
ASSUMING VARIABLE SETTLING VELOCITIES
5000
1000
1000
500
100
50
10
1
Concentrations in ppb

Figure 6-19a. PCB concentrations on bottom due to sinking spill at incineration site, 25 days after spill of entire cargo.

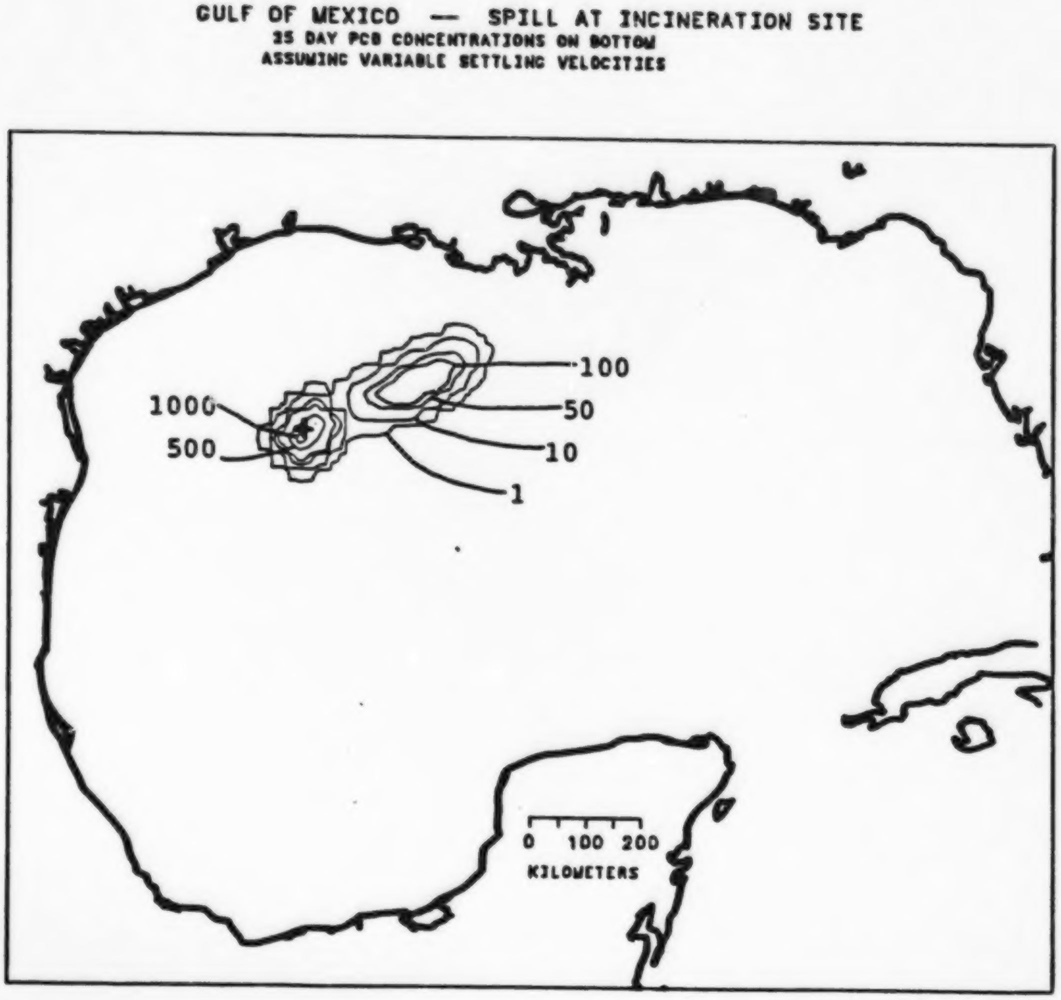

GULF OF MEXICO — SPILL AT INCINERATION SITE
35 DAY PCB CONCENTRATIONS ON BOTTOM
ASSUMING VARIABLE SETTLING VELOCITIES

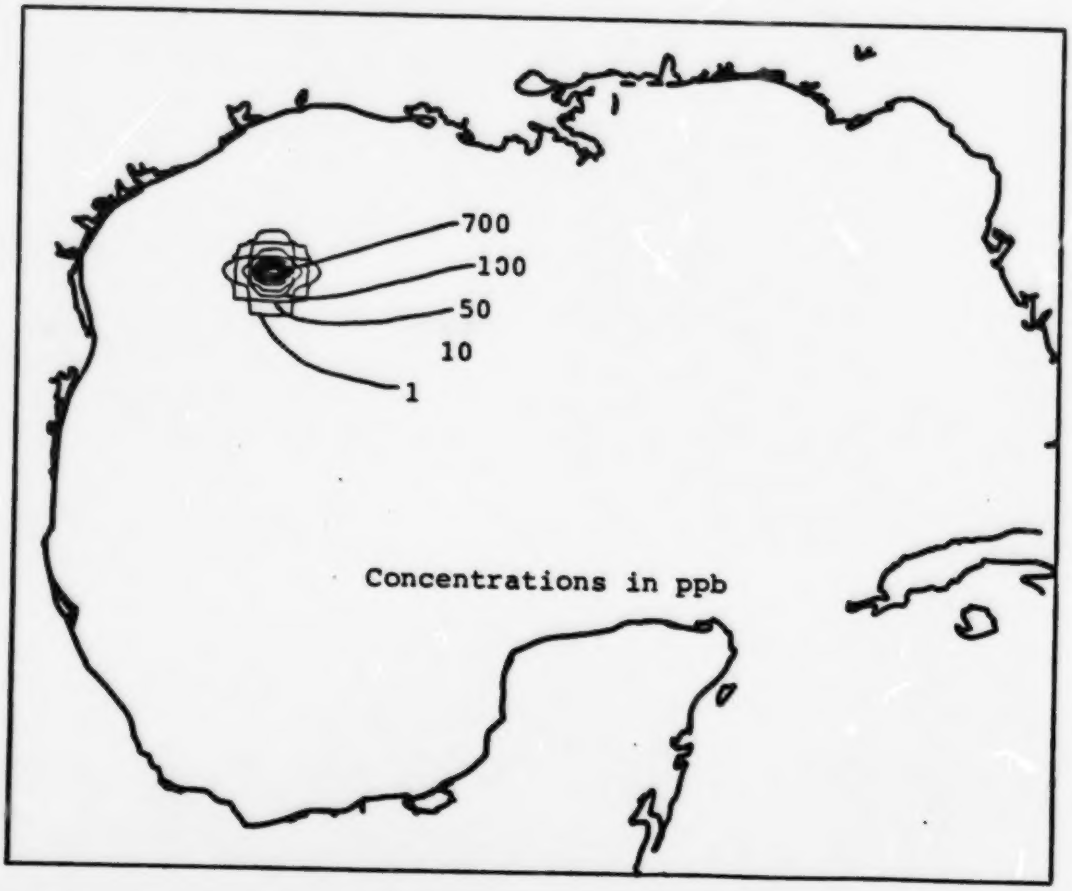
Figure 6-19b. As 6-19a, except for a spill of 1/4 of cargo.

of release. The movement and predicted concentrations of PCB in the top 1 m of the water column are shown in Figures 6-20 through 6-22. Because of weathering and particulate interactions, it is probable that all PCBs would have sunk to the bottom within a month of the spill. Effects of weathering on surface slicks are discussed in Section 6.1.3 and 6.3. Because these effects have been neglected here, the surface layer concentration shown in Figures 6-21 and 6-22 are approximately 20% and 35% too high, respectively.

Response of Offshore Burn Zone Ecosystem to Spills of PCBs

If a spill of PCBs at the offshore burn site sinks, the transport model predicts that the spill would distribute over an area of 5.7×10^{10} m^2. Sediment concentrations initially would average 315.6 ppb over a depth of 5 cm for a 1400 MT spill. Assuming dissolution rates of 1 ppb per day and 0.1 ppb per day as in the shelf case, the ecosystem responses would be as illustrated in Figures 6-23 to 6-26.

For both dissolution rates, PCB levels increase by 3-4 orders of magnitude in the benthos and in fish, while plankton are not affected (Figures 6-23, 6-25), but contamination is lower than in the shelf cases due to greater dispersion of the spill. Benthic biomass declines immediately (to zero) following the spill due to lethal levels of PCB (Figures 6-24, 6-26). Pelagic fish are also reduced due to loss of food resources (mainly demersal fish). The slower loss rate of 0.1 ppb per day causes a longer disturbance of the system, which is long enough to cause



GULF OF MEXICO -- INCINERATION SITE SPILL
10 DAY PCB CONCENTRATIONS AT WATER SURFACE
ASSUMES NO SETTLING OF POLLUTANT

Figure 6-20. PCB concentrations in upper 1 m of water column due to floating spill at incineration site, 10 days after spill.

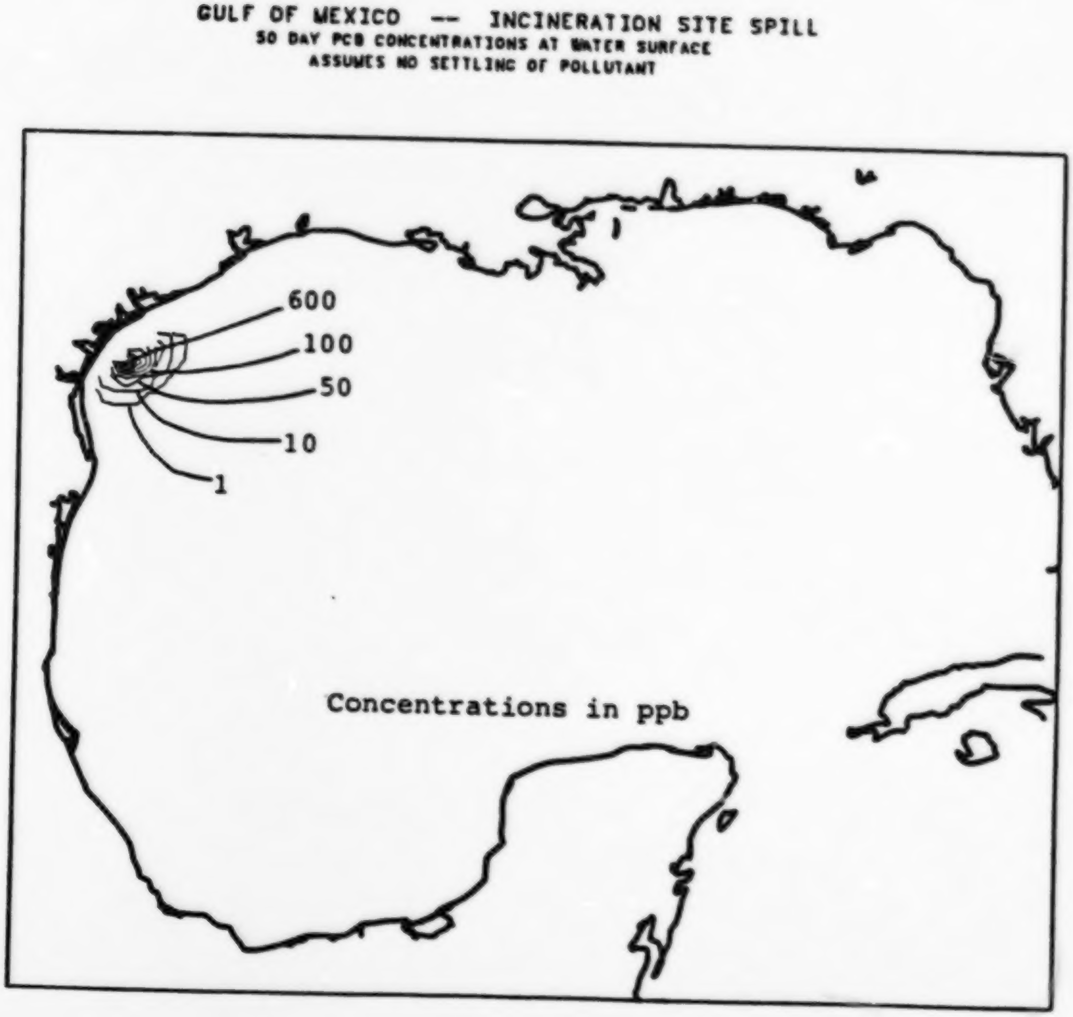

GULF OF MEXICO -- INCINERATION SITE SPILL
50 DAY PCB CONCENTRATIONS AT WATER SURFACE
ASSUMES NO SETTLING OF POLLUTANT

Figure 6-21. PCB concentrations in upper 1 m of water column due to floating spill at incineration site, 50 days after spill.

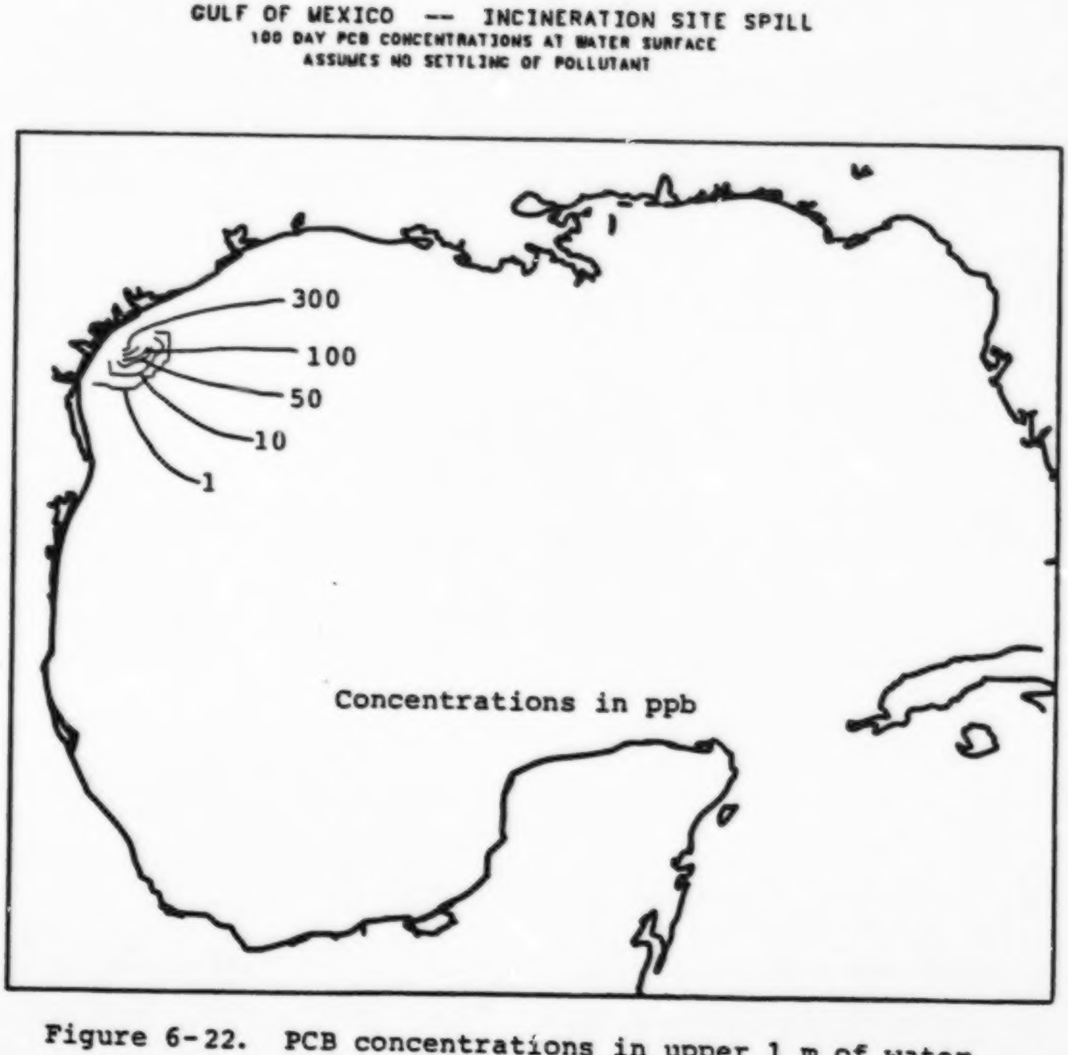

GULF OF MEXICO -- INCINERATION SITE SPILL
100 DAY PCB CONCENTRATIONS AT WATER SURFACE
ASSUMES NO SETTLING OF POLLUTANT

Figure 6-22. PCB concentrations in upper 1 m of water column due to floating spill at incineration site, 100 days after spill.

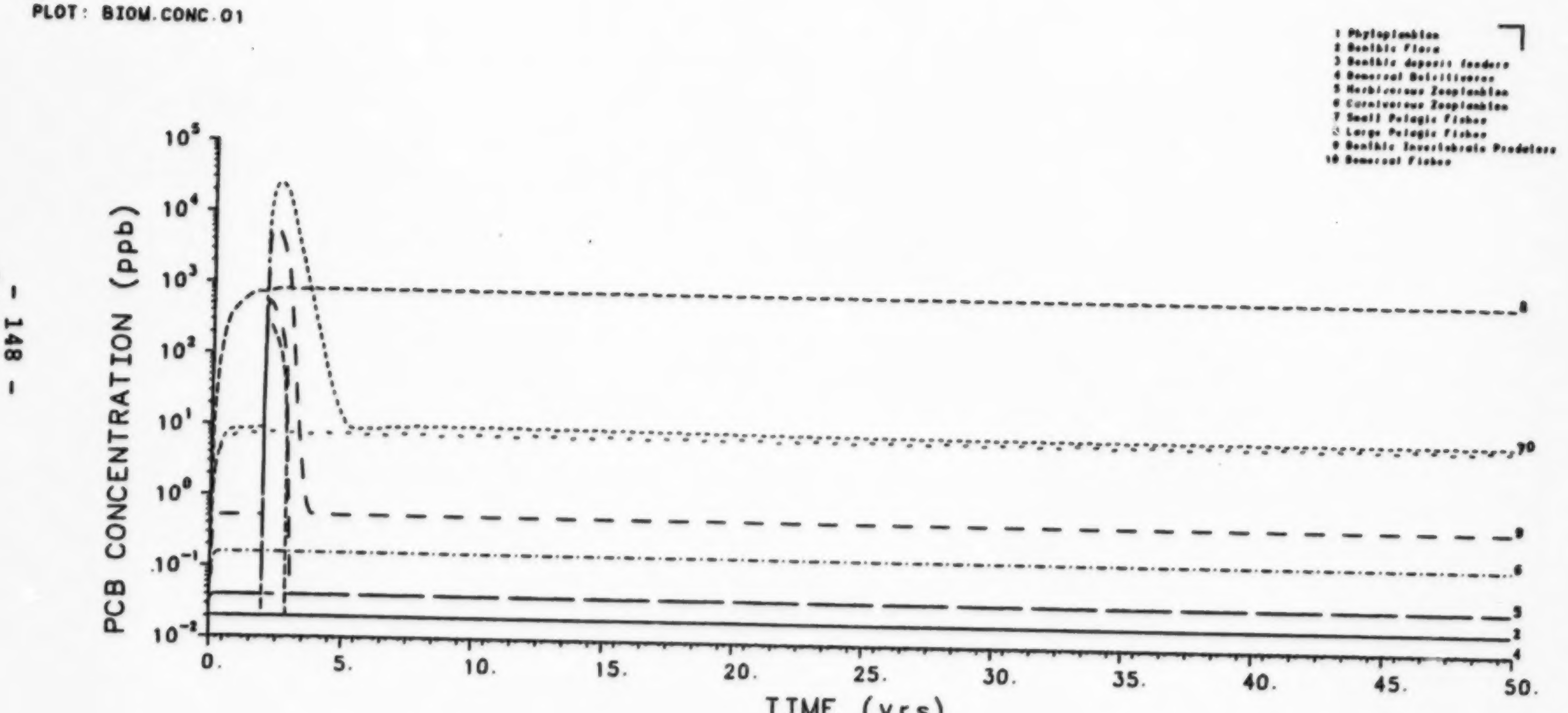


Figure 6-23a. PCB concentrations in trophic compartments of offshore ecosystem before and after a spill of the entire cargo at two years at the offshore burn zone. Loss rate from sediment is 1.0 ppb/day.

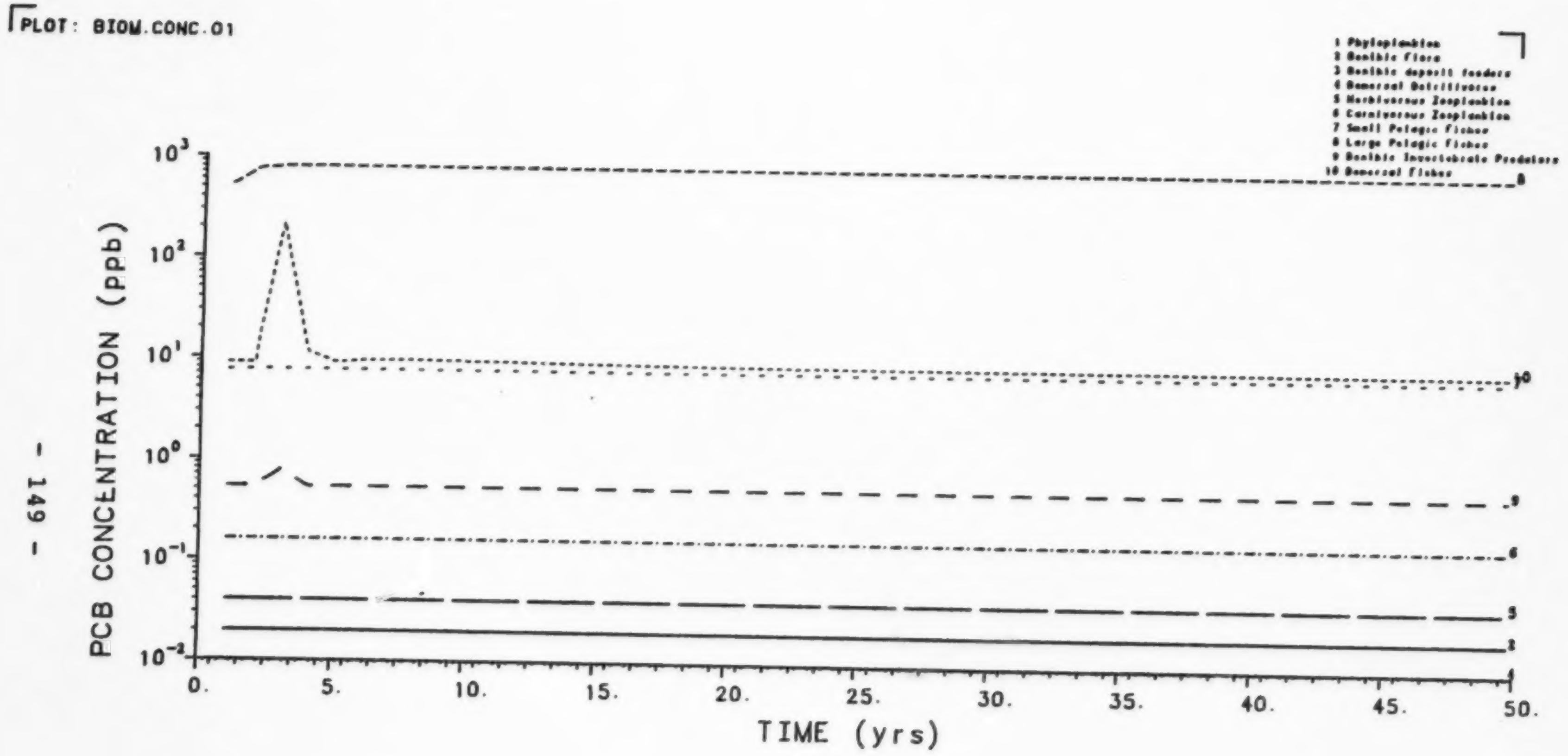


Figure 6-23b. As 6-23a, except 1/4 the cargo is spilled.

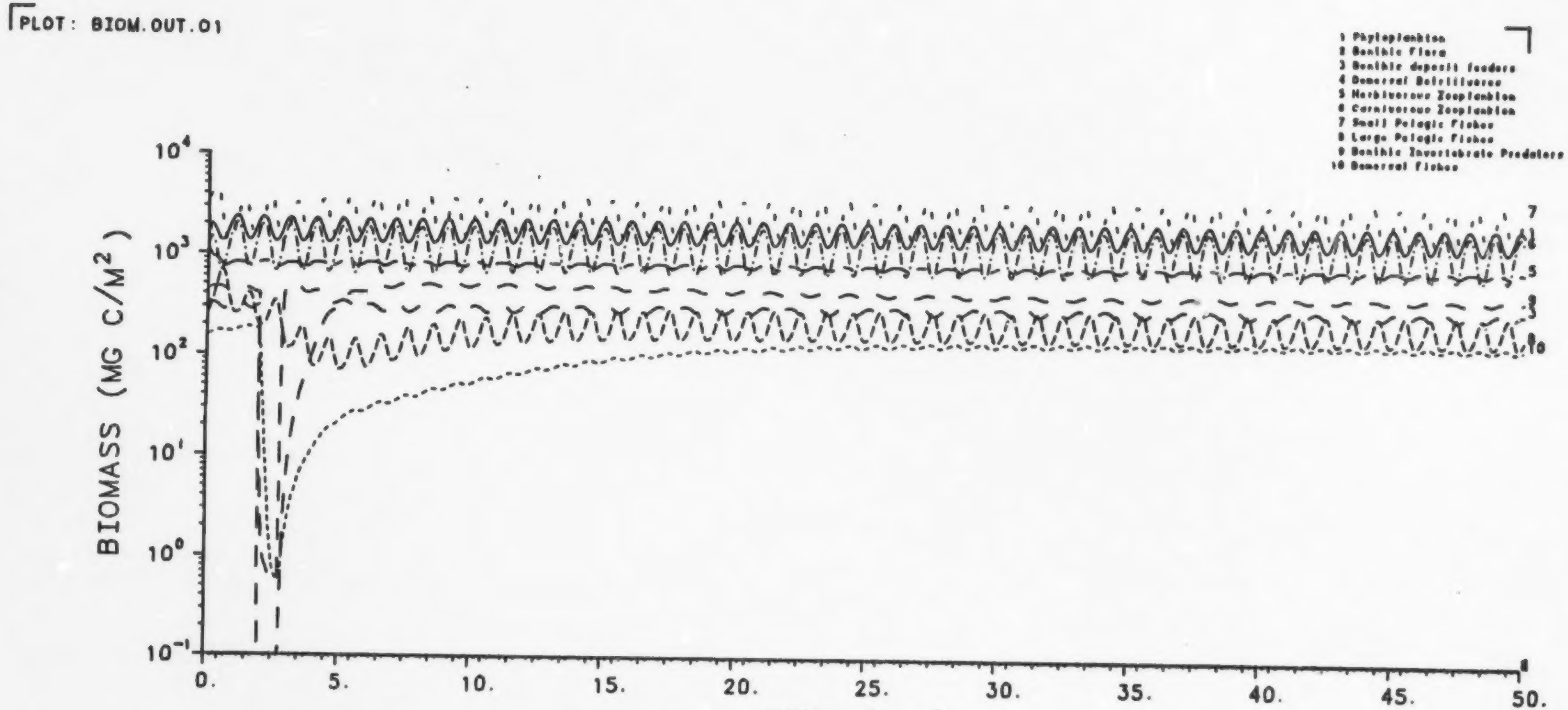


Figure 6-24a. Biomass levels before and after a spill of the entire cargo at two years at the offshore burn zone. Loss rate from sediment is 1.0 ppb/day.

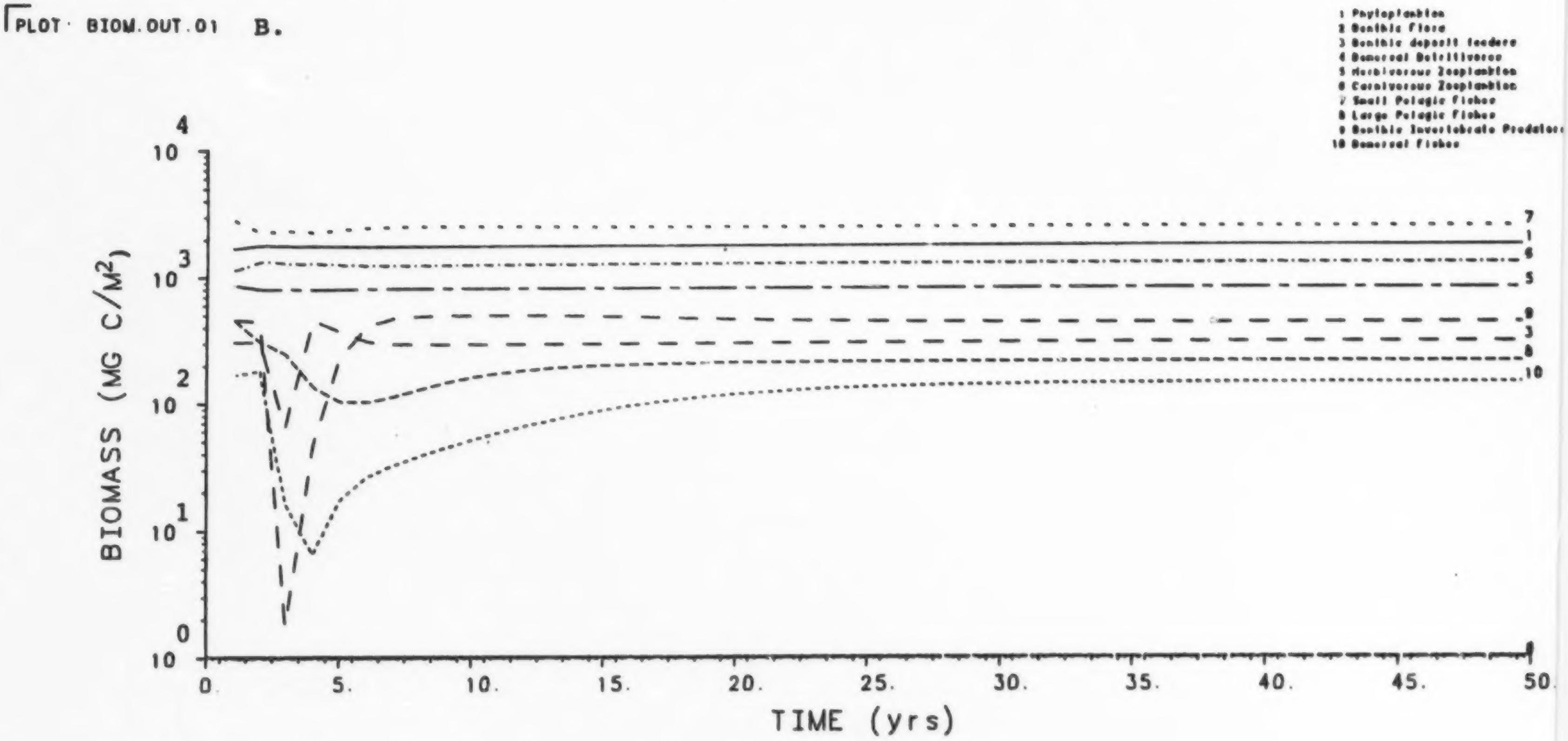


Figure 6-24b. As 6-24a, except yearly average of biomass is plotted versus time.

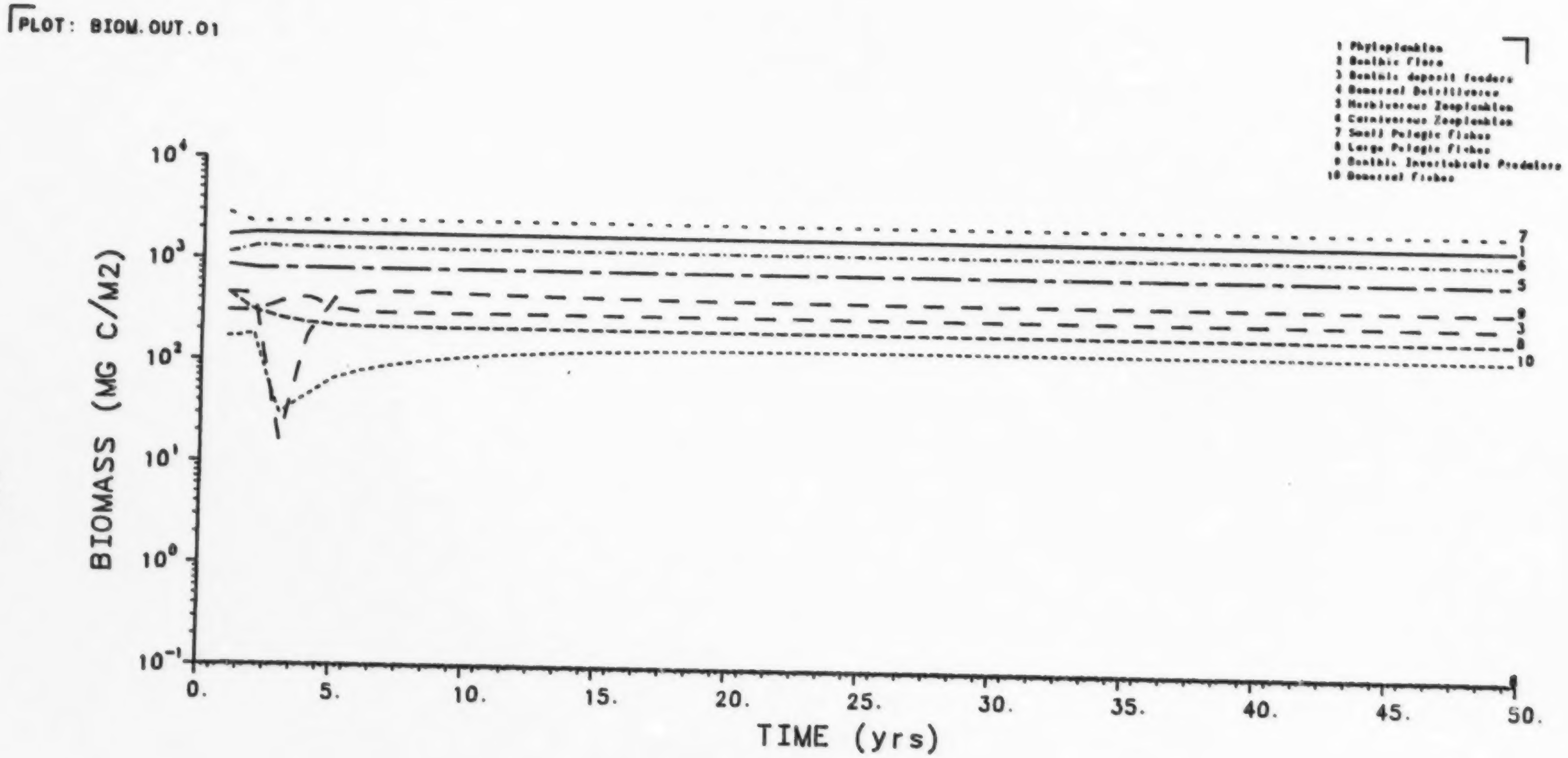


Figure 6-24c. As 6-23b, except 1/4 of the cargo is spilled.

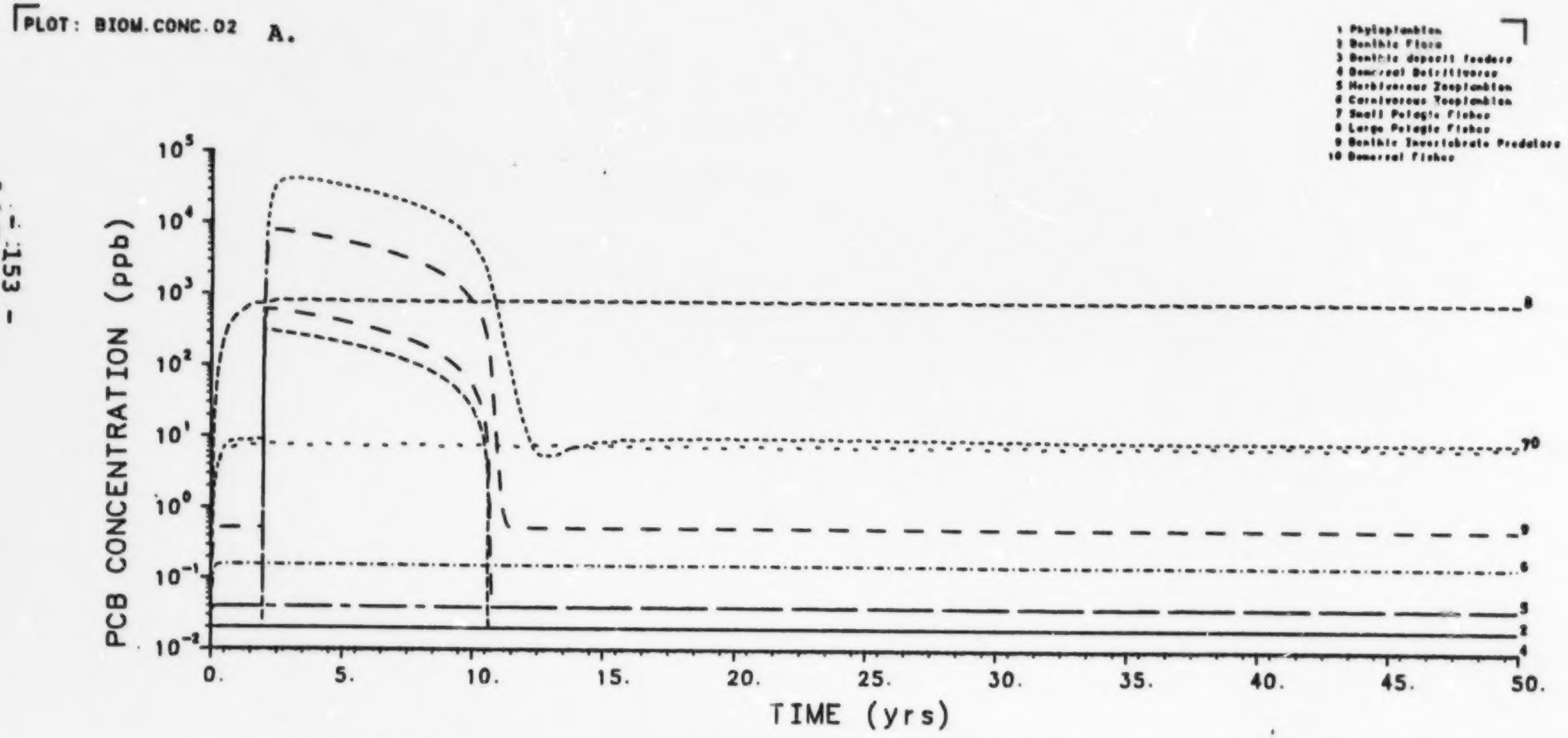


Figure 6-25a. Concentrations of PCB's in trophic compartments of offshore ecosystem before and after a spill of the entire cargo at two years at the offshore burn zone. Loss rate from sediment is 0.1 ppb/day.

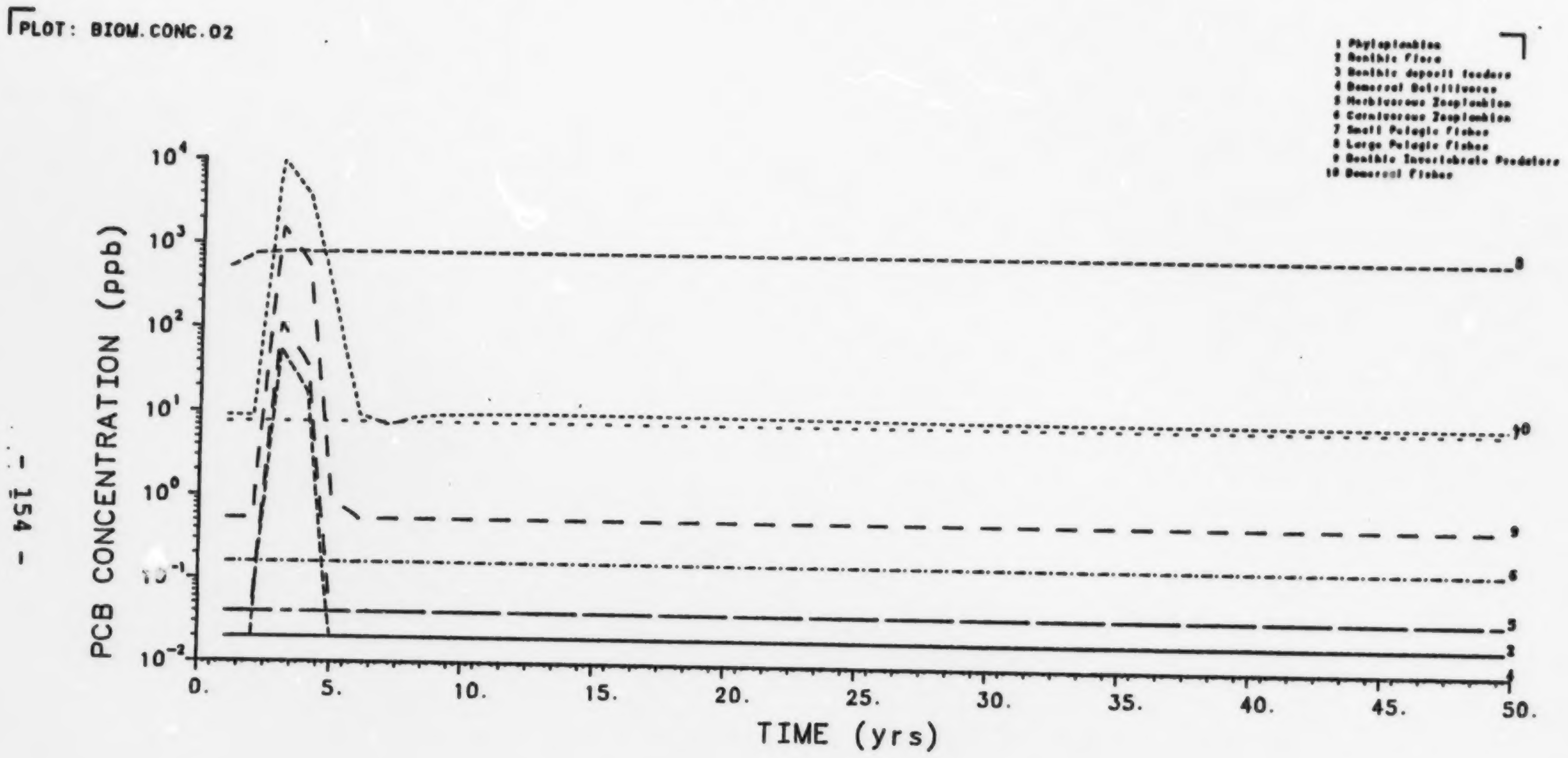


Figure 6-25b. As 6-25a, except 1/4 of the cargo is spilled.

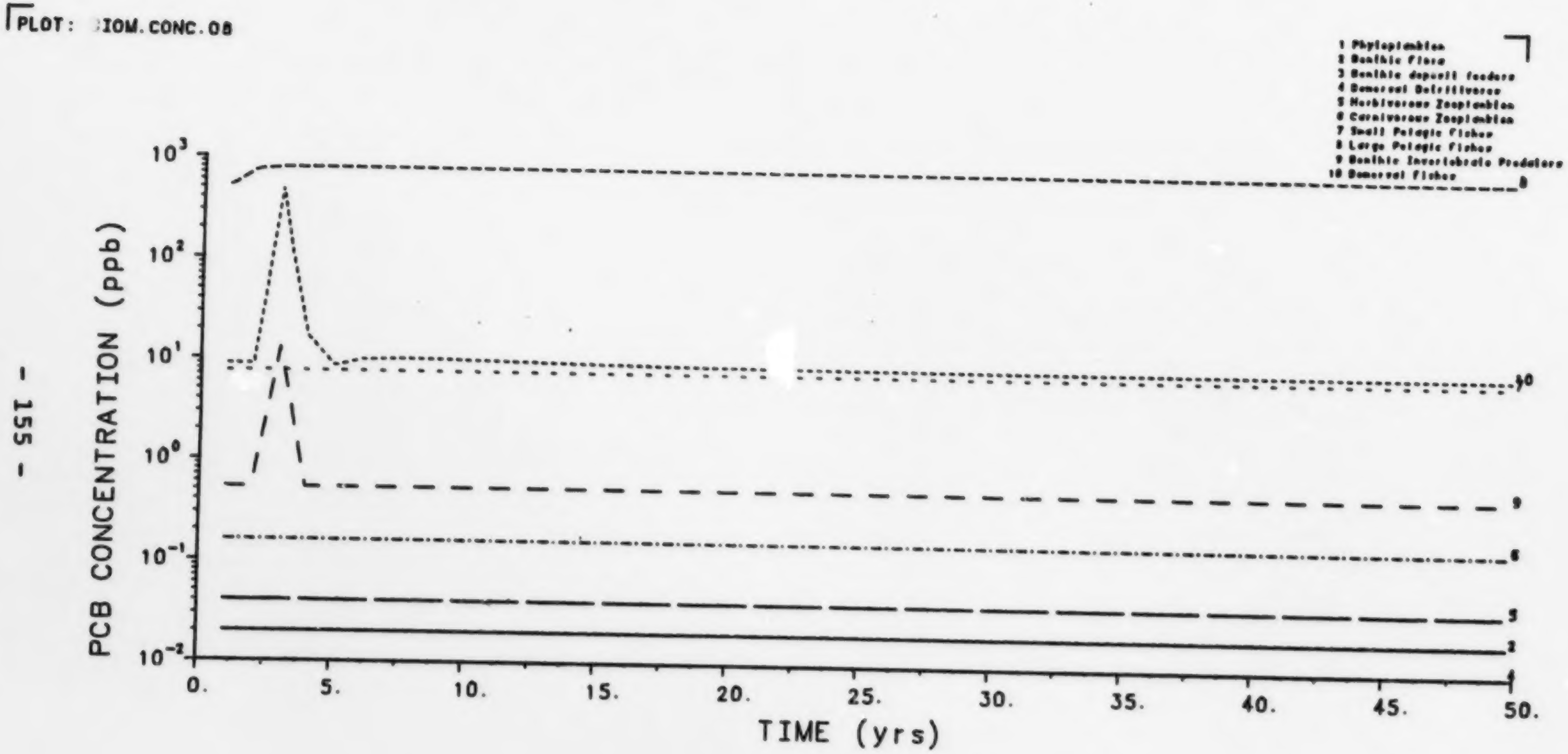


Figure 6-25c. As 6-25a, except 1/16 of the cargo (1/2 tank) is spilled.

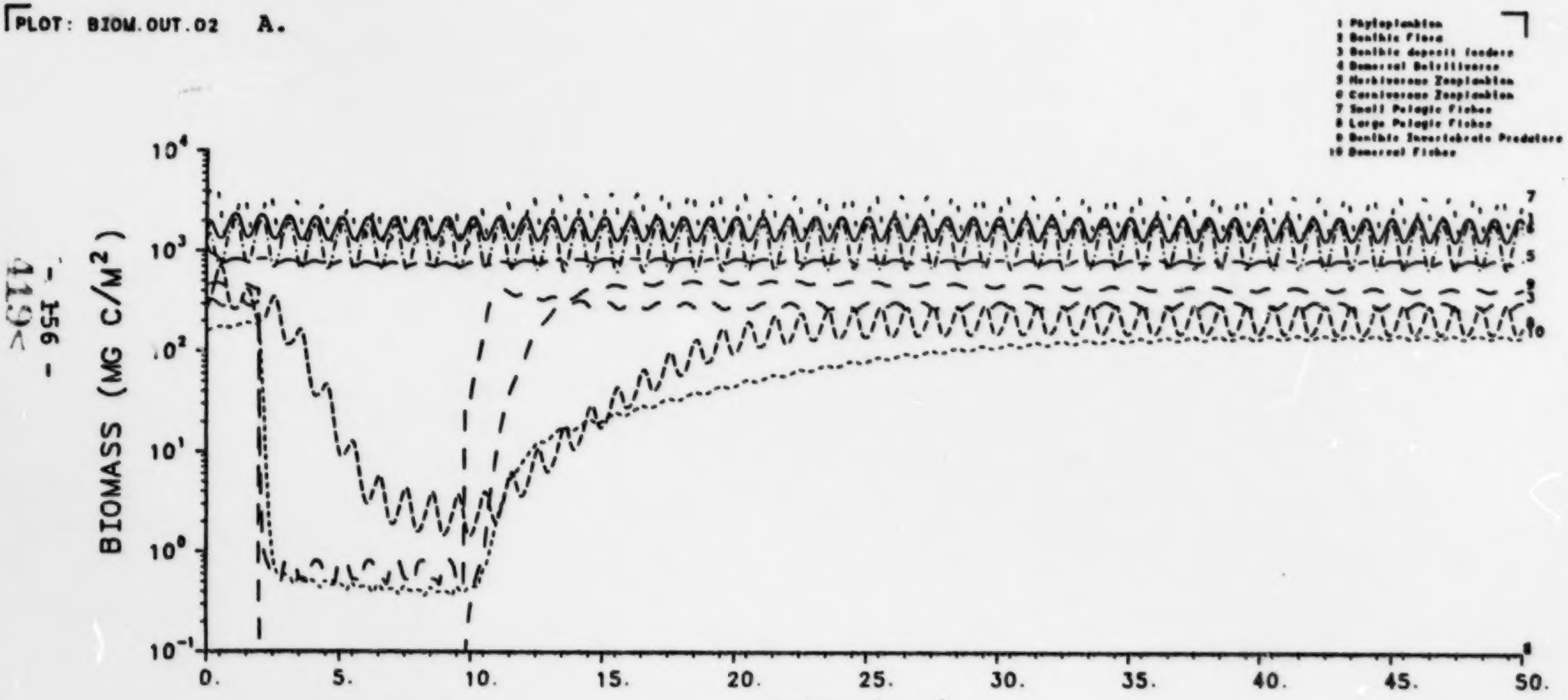


Figure 6-26a. Biomass levels before and after a spill of the entire cargo of PCB's at two years at the offshore burn zone. Loss rate from sediment is 0.1 ppb/day.

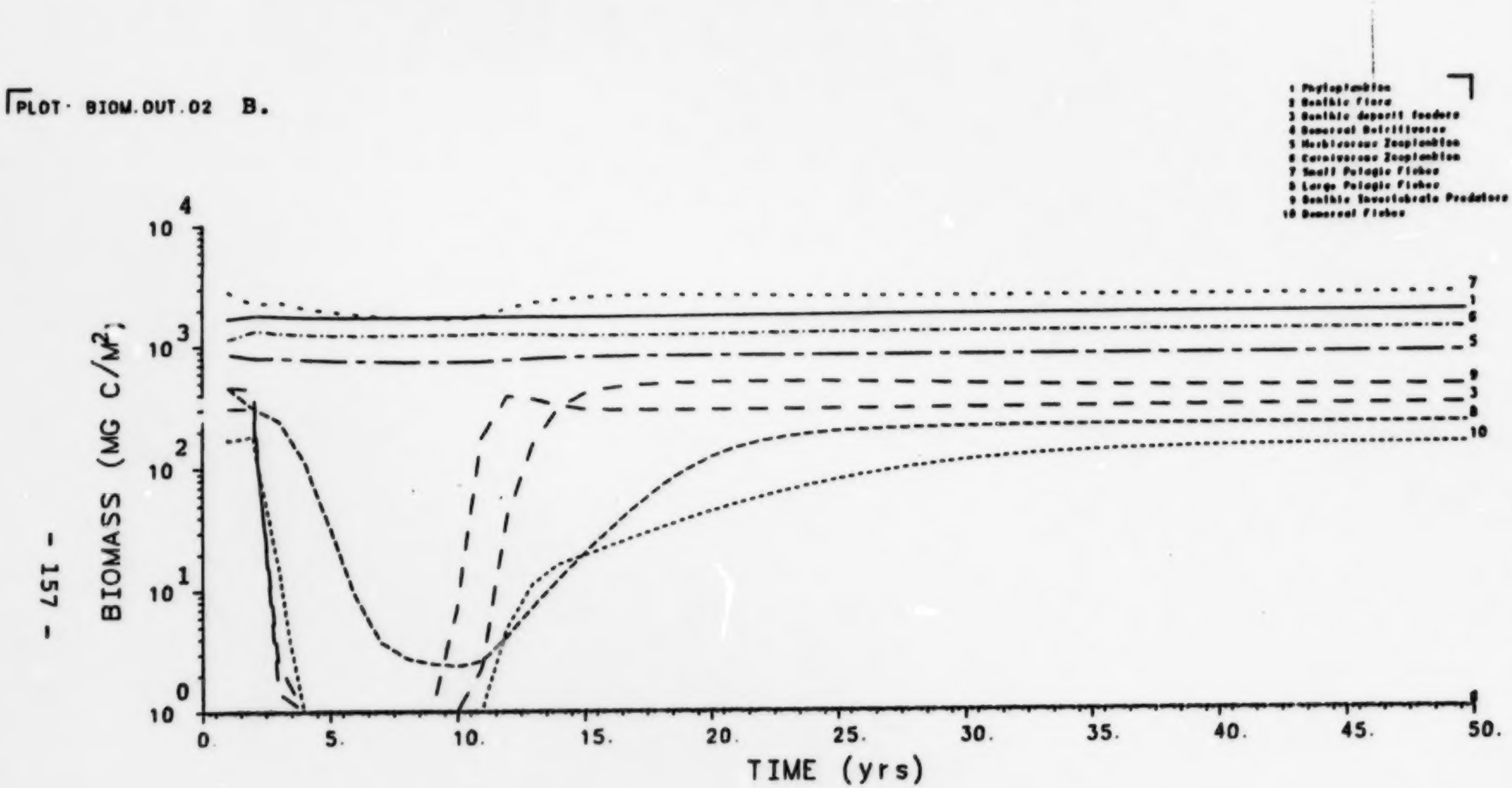


Figure 6-26b. As 6-26a, except yearly average of biomass is plotted versus time.

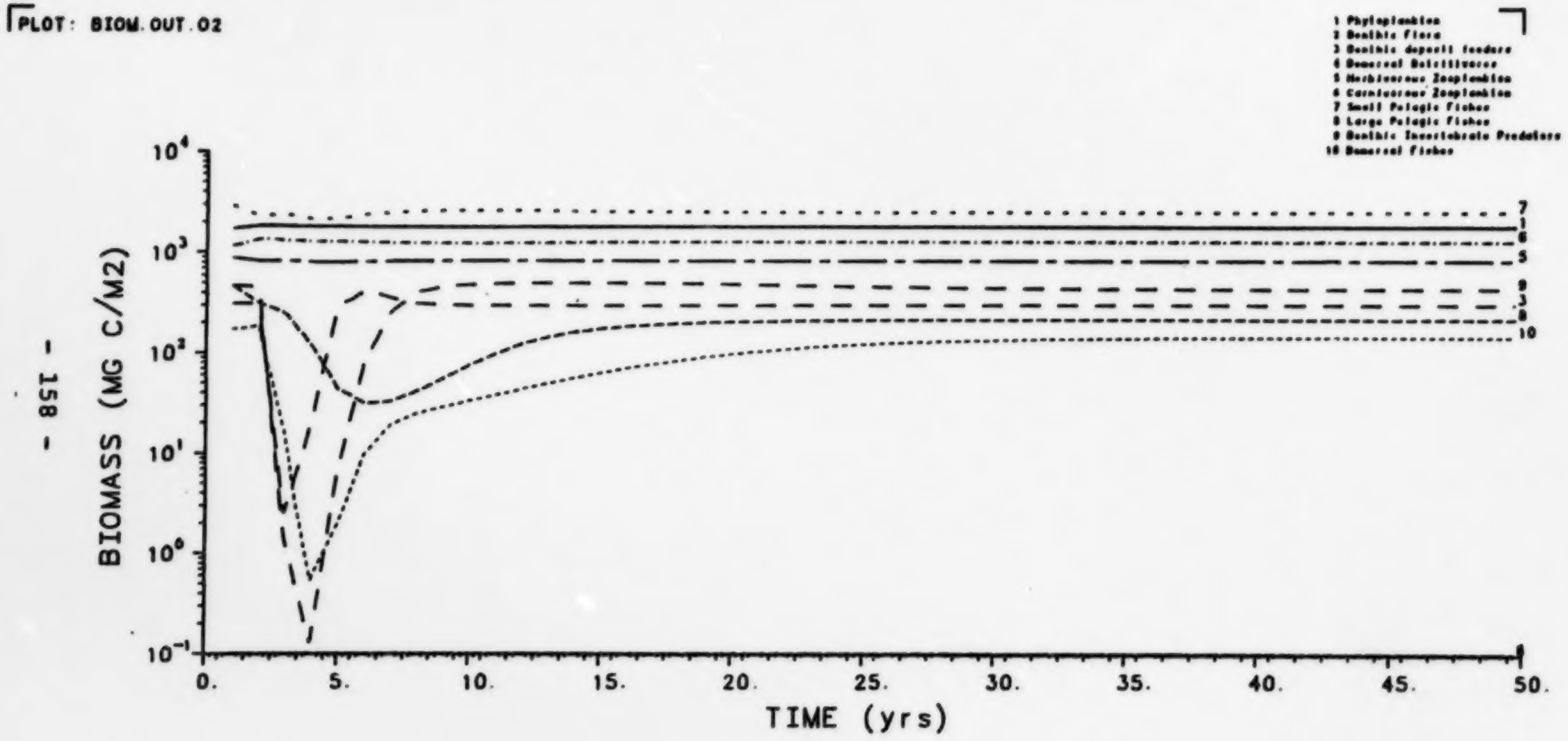


Figure 6-26c. As 6-26b, except 1/4 the cargo is spilled.

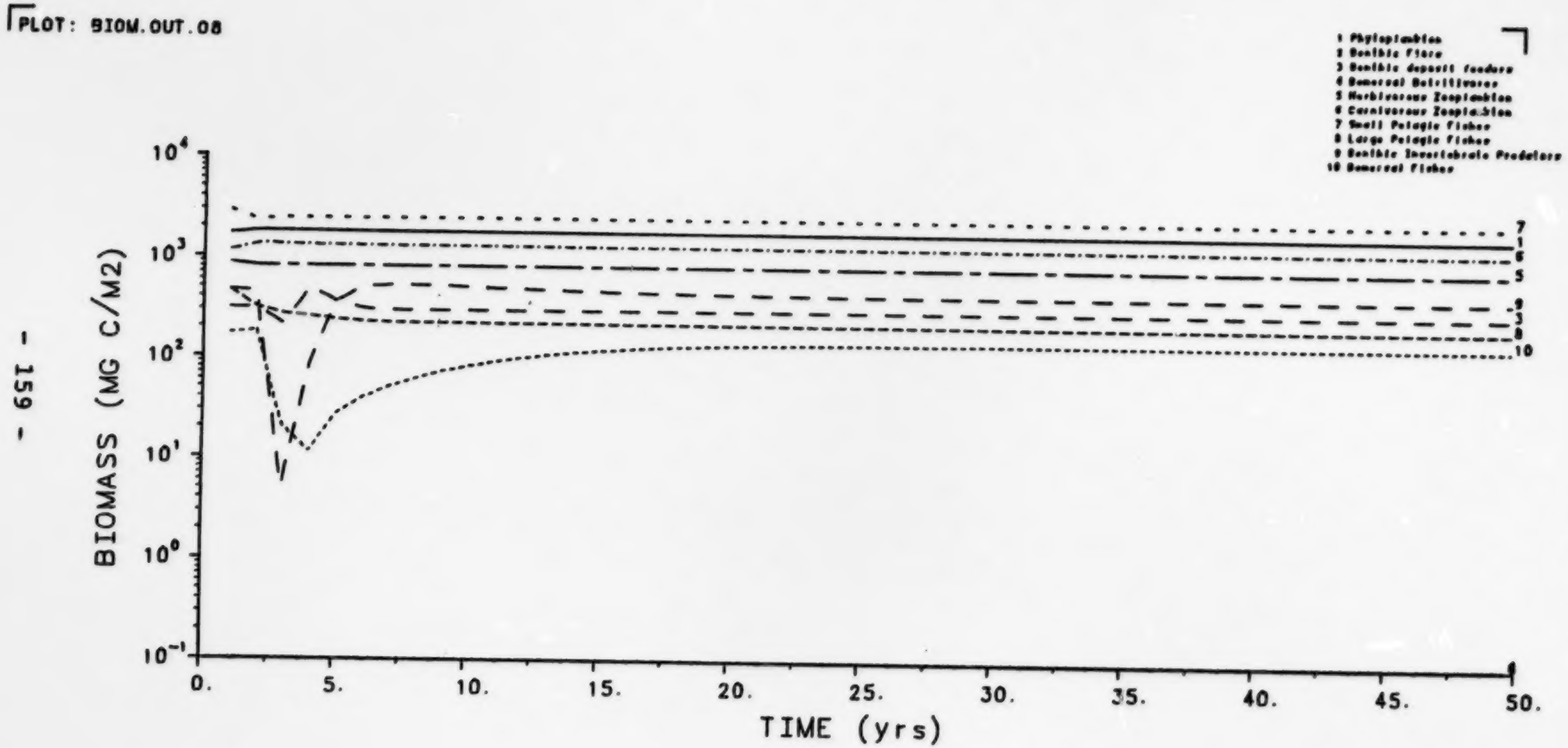


Figure 6-26d. As 6-26b, except 1/16 of the cargo (1/2 tank) is spilled.

local extinction of large pelagic fish (#8) due to significant loss of food resources (Figure 6-26).

Assuming immigration occurs, the benthos and fish recover once PCBs have dissolved and disappeared from the sediments. In the first case (1 ppb per day loss rate, Figure 6-23, 6-24), the sediments would return to background levels after one year, but recovery of the ecosystem is not complete for nearly 20 years. In the case of a loss rate of 0.1 ppb per day (Figures 6-25, 6-26), sediments would return to background level of PCBs after 8 years, and complete ecosystem recovery requires 20 years beyond that point. The demersal fish are slowest to recover while benthic invertebrates recover quickly due to their faster growth rates. As mentioned earlier, these loss rates are fairly arbitrary and there is substantial uncertainty concerning the likely duration of effects.

For a smaller spill of only 350 MT of PCB, and assuming a loss rate of 1 ppb per day, the effect on the ecosystem is slight both in terms of body burdens and biomass response (Figures 6-23b, 6-24c). The initial sediment concentrations would be only 90 ppb, and so the contamination would disappear by the end of 3 months. If the sediment loss rate is on the order of 0.1 ppb per day, however the body burdens and biomass levels would initially respond with the same pattern as the full cargo spill, but the duration of the response would be 1/4 the time (Figure 6-25, 6-26), as in the shelf case. Thus, assessment of impact on the ecosystem is very sensitive to the magnitude of the loss rate from the sediments.

If only 1/2 a tank (1/16 of the cargo) is spilled at the burn site, the sediment concentration in the contaminated area would average 29 ppb. Even at the slower dispersal rate of 0.1 ppb per day, the effect on the ecosystem is relatively minor (Figures 6-25c and 6-26d). Only benthic predators and demersal fish would show increased levels of contamination, on the order of 40 times pre-spill levels. These two compartments would decline in biomass by an order of magnitude due to lowered food supply, but would recover relatively rapidly following the spill. Immigration would further speed up this recovery.

6.2 EDC Spills

6.2.1 Ecosystem Model Predictions at Background Levels of EDC

The chemical 1,2-Dichloroethane is a member of the family of chlorinated ethanes. Other names include Ethylene Dichloride, Ethylene Chloride, and the acronym, EDC. Background levels of EDC have not been found in the literature for the Gulf of Mexico, but are assumed to be on the order of 1 ppb. This level is well below toxic levels for marine organisms (Figure 4-3)

Table 6-2 gives steady state background EDC levels in each trophic compartment assuming assimilation and depuration rates are as listed. EDC is not concentrated in the lower trophic levels and is bioconcentrated to a factor of about 2 in fish, in agreement with data provided by Veith et al. (1980). The simulations give nearly identical results for the three regions, the only variation being at the top of the food chain. Demersal

Table 6-2. Predicted EDC levels (ppb) by compartment and site when water and sediment concentrations are held at a background level of 1.0 ppb EDC.

#	Compartment	Asimilation Rate	Depuration Rate (Day^{-1})	Mobile Bay	Shelf	Offshore
3	benthic deposit feeders	.40	.40	1.0	1.0	1.0
4	demersal detritivores	.40	.40	1.0	1.0	--
5	herbivorous zooplankton	.40	.40	1.0	1.0	1.0
6	carnivorous zooplankton	.40	.40	1.0	1.0	1.0
7	small pelagic fish	.50	.30	1.7	1.7	1.7
8	large pelagic fish	.50	.30	--	2.3	2.6
9	benthic invertebrate predators	.40	.40	1.0	1.0	1.0
10	demersal fish	.50	.30	2.5	1.7	1.7

fish increase in EDC concentration onshore due to larger portions of their diet being made up of pelagic organisms in shallower regions.

6.2.2 Mobile Bay

Estimating Mobile Bay Instream EDC Concentrations

In-stream concentrations resulting from a spill of 3800 metric tons of waste with a 50% EDC content were calculated based on the methodology of Section 6.1.2. The dissolution rate is a strong function of the difference between the solubility of the compound and ambient concentrations. EDC solubility in water is on the order of 8.1 g/l (U.S. EPA, 1980). Given the extreme solubility of the chemical (almost 700,000 times that of Auroclor 1254), it can be shown that the total spill of EDC would probably be in solution within several hours. In-stream values for this chemical can therefore be generated from a simple residence time calculation (Fig. 6-27).

The initial in-stream concentration can be derived by dividing the mass of EDC spilled (1.9×10^6 kg) by the volume of Mobile Bay (2700×10^6 m^3. The value is approximately 700 ug/l. Flushing 12.4% of the Bay waters each day with tidal and river inflows results in the EDC concentrations of Mobile Bay returning to the pre-spill background value of 1.0 ug/l in about 60 days. As for the PCB concentration calculations in this shallow area, neglecting volatilization results in a high residence time estimate. Using methods described in Lyman et

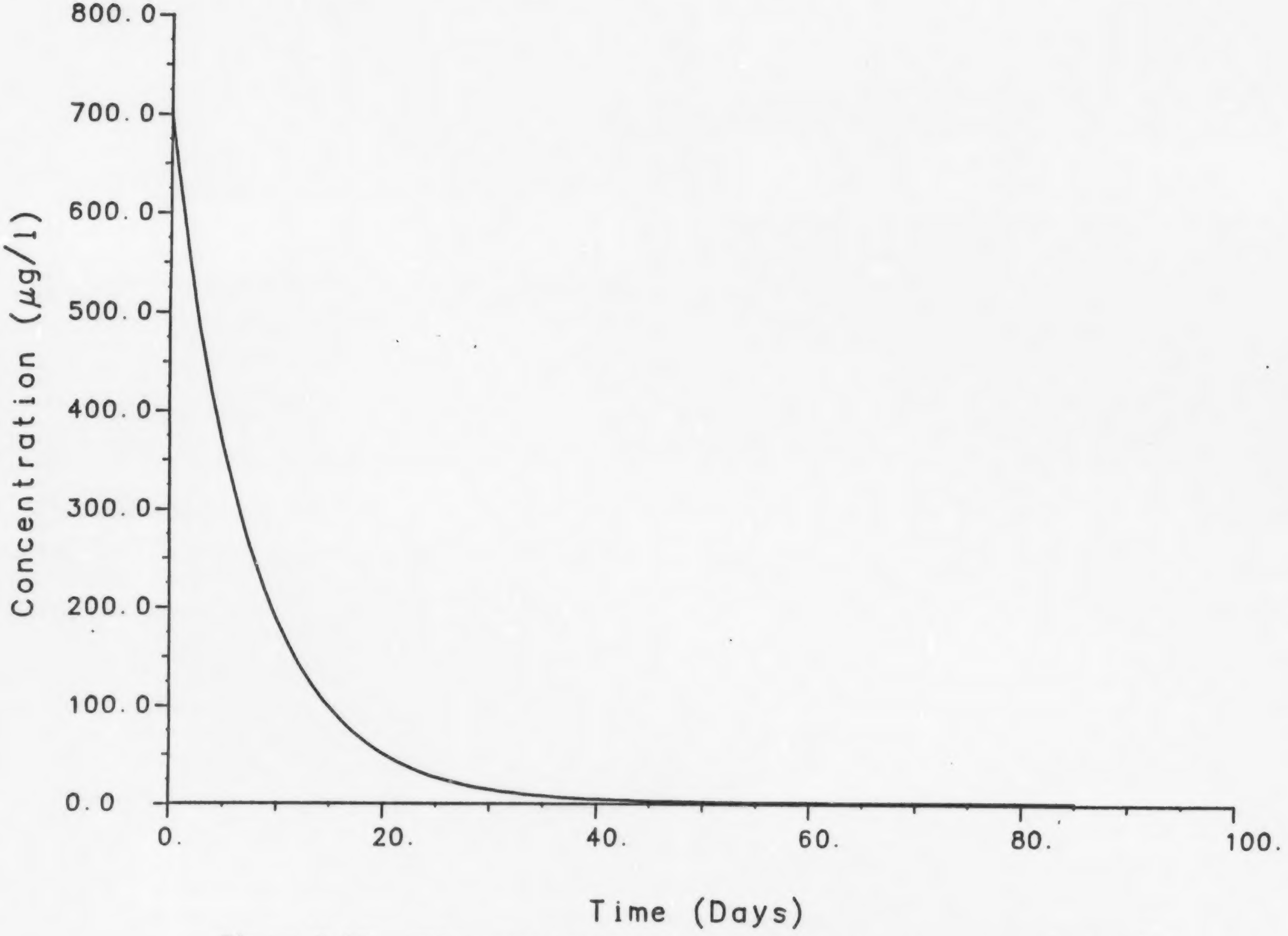


Figure 6-27. EDC concentrations in the water as a function of time for,

al. (1982), the half life of EDC in 3 meters of water is less than one day. Thus volatilization is several times more rapid than flushing for this chemical in this environment, if the chemical is 100% dissolved.

Response of Mobile Bay System to Spills of EDC

Figures 6-28 and 6-29 illustrate the effect of a spill of EDC at two years. EDC concentrations increase in all trophic levels immediately following the spill, but quickly decrease and return to background levels when the EDC is flushed from the bay 40 days later. Since the levels of EDC in the water never reach toxic concentrations (Figure 4-3), biomass levels are not altered by the spill in the simulation (Figure 6-29).

6.2.3 Continental Shelf Spill Scenario

Pollutant Transport Model Predictions

The two-layer transport model described in Section 2 and used to simulate the PCB spill in Section 6.1 was also used to simulate an accidental release of EDC on the shelf. The spill location is the same as that used previously, at 29 degrees 40 minutes N, 88 degrees 30 minutes W, along the ship's track to the incineration site. The simulation assumes the entire cargo of the ship to be released at once. Assuming roughly 50% of the 3800 metric ton cargo to be comprised of EDC, 1900 metric tons of EDC are spilled. Since EDC is highly volatile and soluble, only one scenario was run. This scenario assumes the EDC to be

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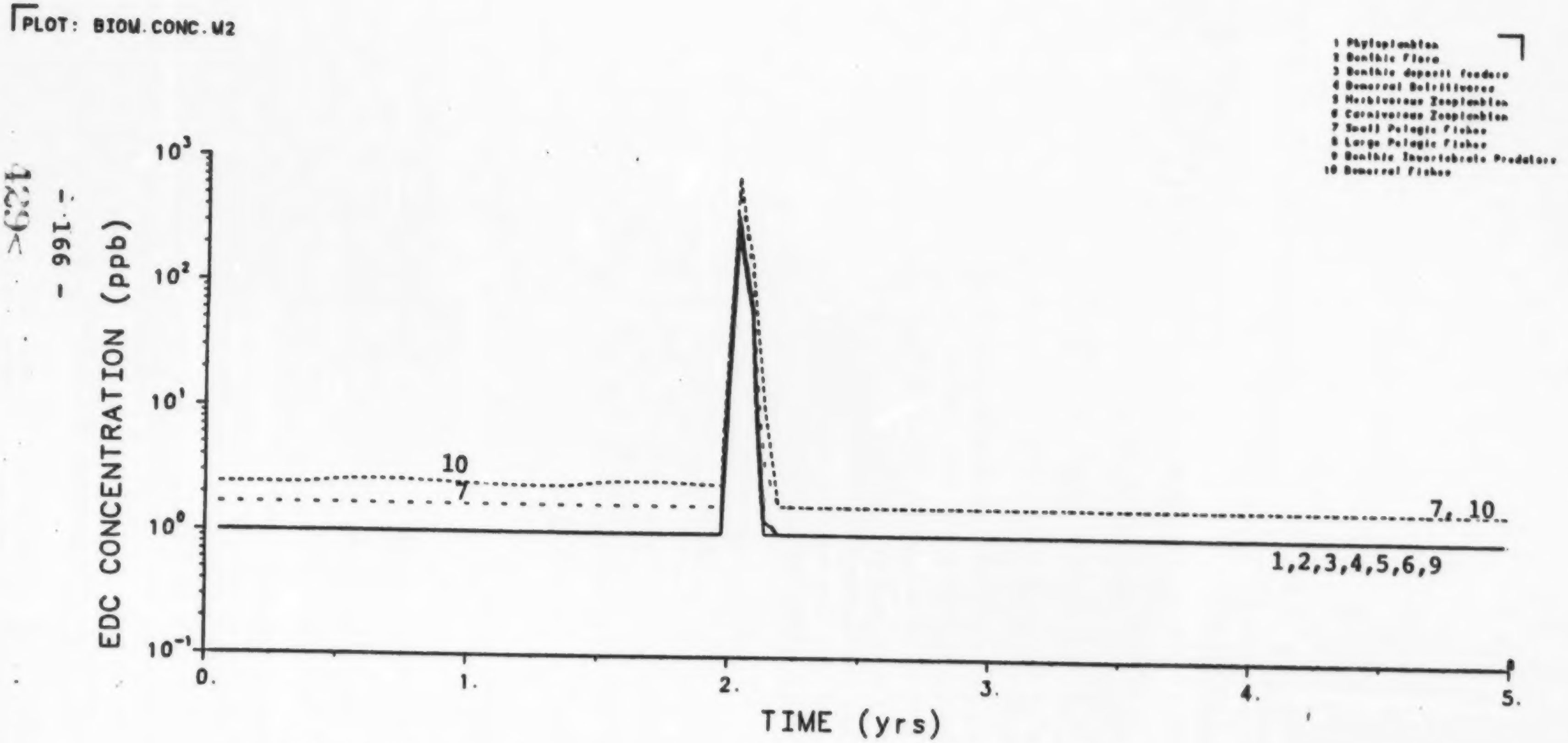


Figure 6-28. Concentrations of EDC by trophic level before and after a spill at 2 years in Mobile Bay.

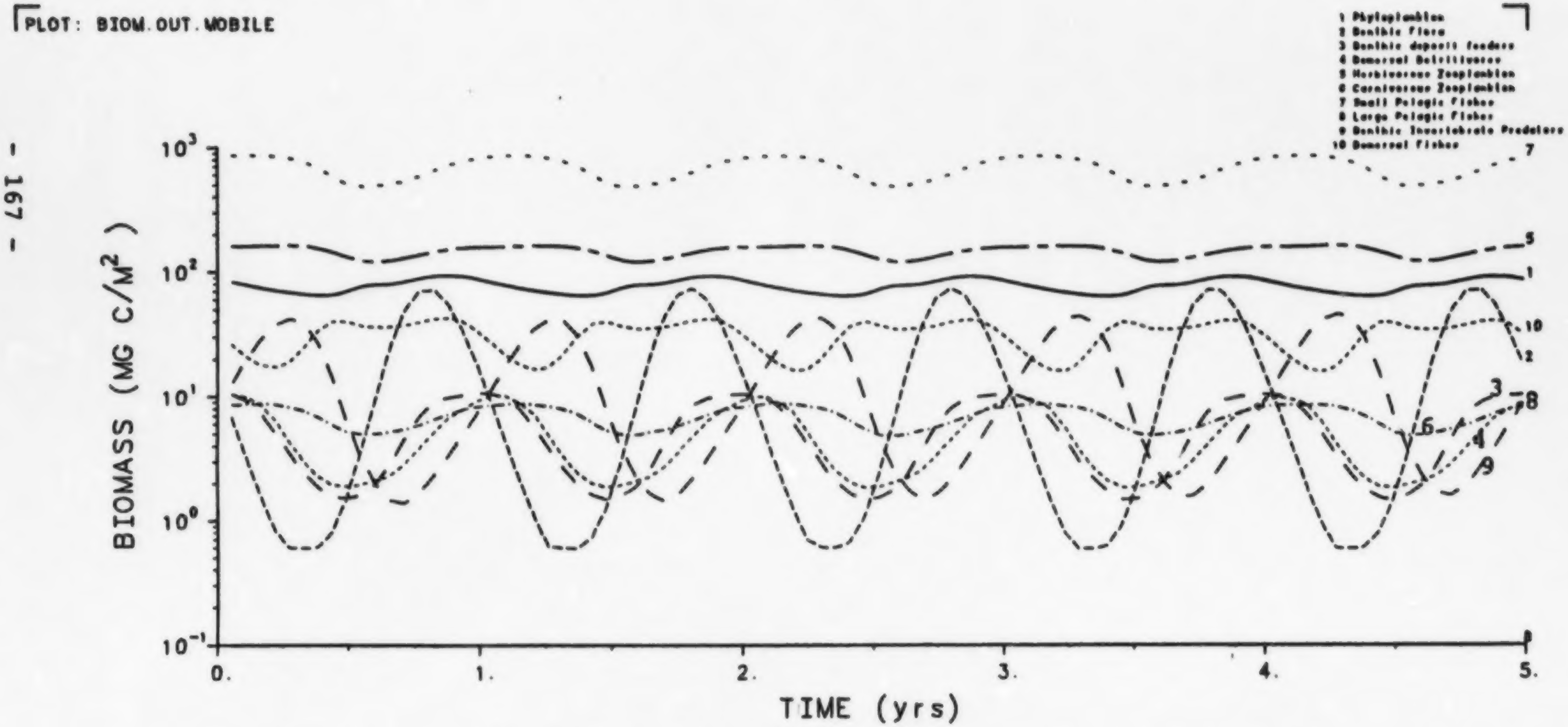


Figure 6-29. Biomass levels before and after spill of EDC in Mobile Bay at two years.

completely dissolved (neutrally buoyant). Therefore its only vertical motion is due to vertical diffusion. Advection by surface and subsurface currents determines the movement of EDC in the horizontal plane.

Figure 6-30 shows the distribution of EDC in the surface layer at 20 days after the spill. Because water depths in this area are generally less than 60 m, subsurface concentrations are not calculated. The EDC concentrations can be assumed to decrease as diffusion and dissolution continue. Evaporation will also contribute significantly to water column concentrations, although for the present "worst case" scenario this effect has been neglected.

Response of Shelf System to Spills of EDC

According to the hydrodynamic model, spills of EDC would disperse rapidly due to advection and the high solubility of EDC in water. After 20 days, maximum concentrations of EDC in the surface and subsurface layers would be 75 ppb and <1 ppb EDC, respectively. These concentrations are well below toxic levels for marine organisms (Figure 4-3). Thus, a spill of EDC along the transport route to the burn zone (on the shelf) is unlikely to have a noticeable effect on the ecosystem.

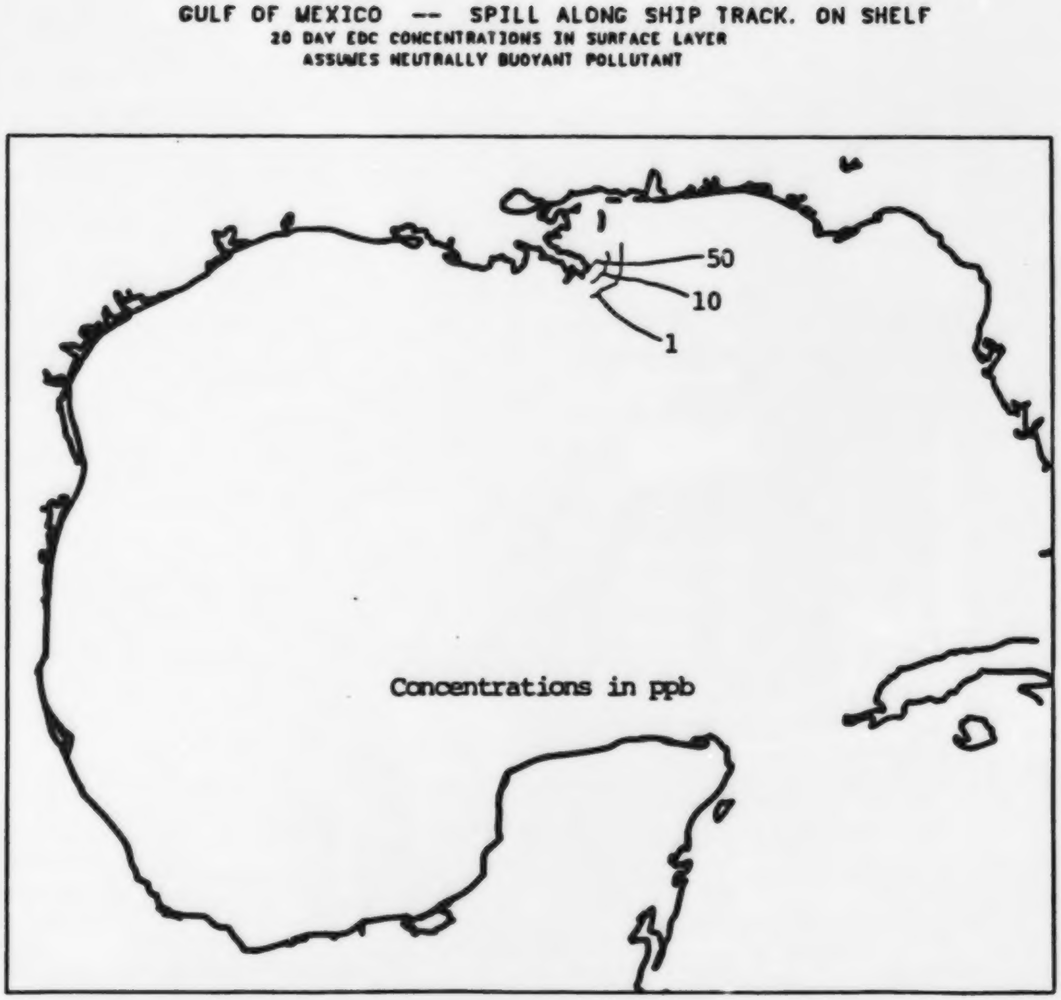

GULF OF MEXICO -- SPILL ALONG SHIP TRACK. ON SHELF
20 DAY EDC CONCENTRATIONS IN SURFACE LAYER
ASSUMES NEUTRALLY BUOYANT POLLUTANT
50
10
1
Concentrations in ppb

Figure 6-30. EDC concentrations in surface layer due to spill on shelf, 20 days after spill.

6.2.4 Burn Zone Spill Scenario

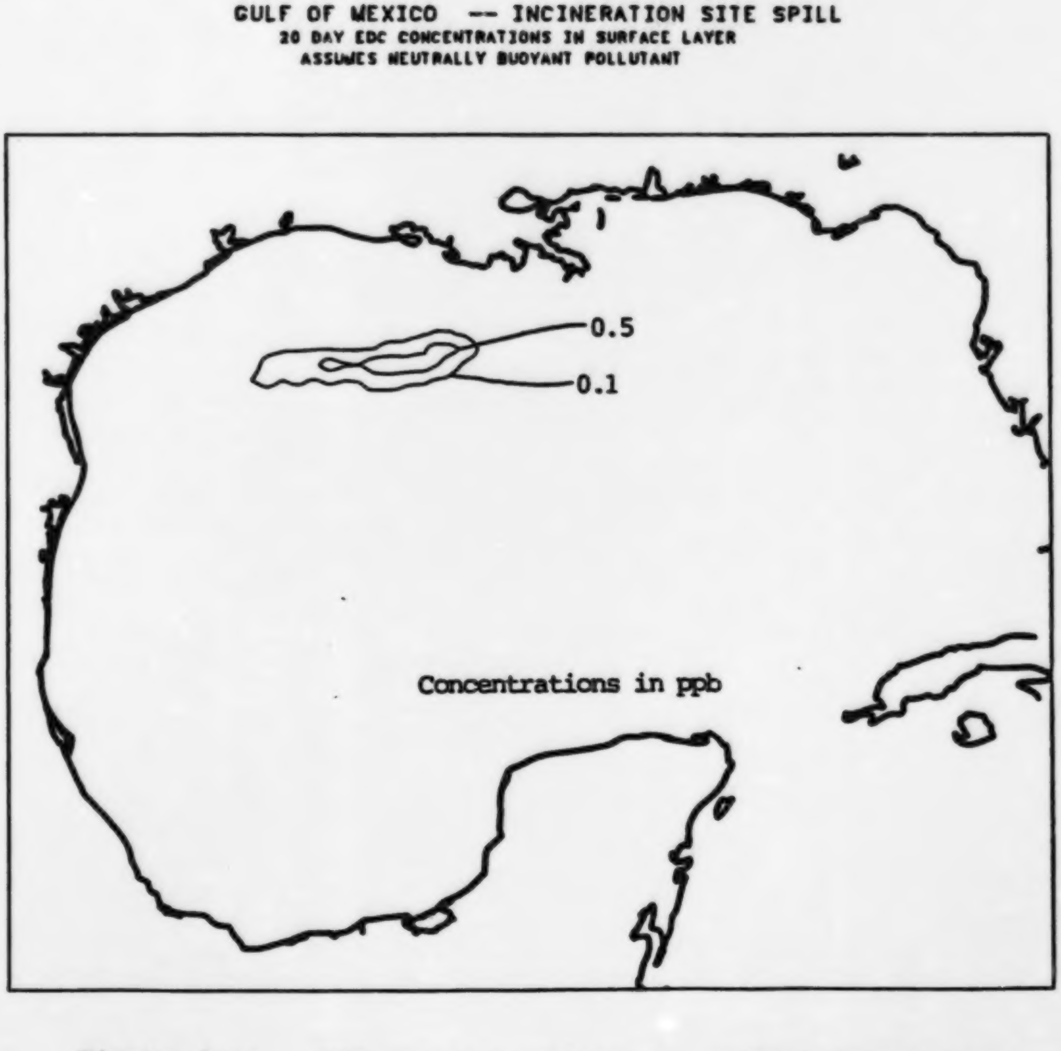
Pollutant Transport Model Predictions

The scenario described in Section 6.2.3 for an accidental release of EDC at a shelf site was repeated at the incineration site. This site, used previously for the PCB spills, is located at 26 degrees 40 minutes N, 93 degrees 40 minutes W.

As before, EDC is treated as a dissolved substance. Figure 6-31 shows the surface concentrations of EDC averaged over the upper 60 m of the water column. Concentrations in the lower water column are shown in Figure 6-32. As can be seen from these figures, EDC concentrations are below background levels by the end of 20 days and do not hit any shoreline. Again, evaporation has been neglected to affect a "worst case" scenario.

Response of Offshore Burn Zone System to Spills of EDC

According to the transport model, maximum concentrations of EDC following a spill would be on the order of 1 ppb in the water column. These concentrations are similar to assumed background levels and are not acutely toxic to marine organisms. Thus a spill of EDC at the burn site should have no acute effect on the ecosystem. The short half life of EDC in organic tissues also precludes bioaccumulation effects.



GULF OF MEXICO -- INCINERATION SITE SPILL
20 DAY EDC CONCENTRATIONS IN SURFACE LAYER
ASSUMES NEUTRALLY BUOYANT POLLUTANT

Figure 6-31. EDC concentrations in surface layer due to spill at incineration site, 20 days after spill.

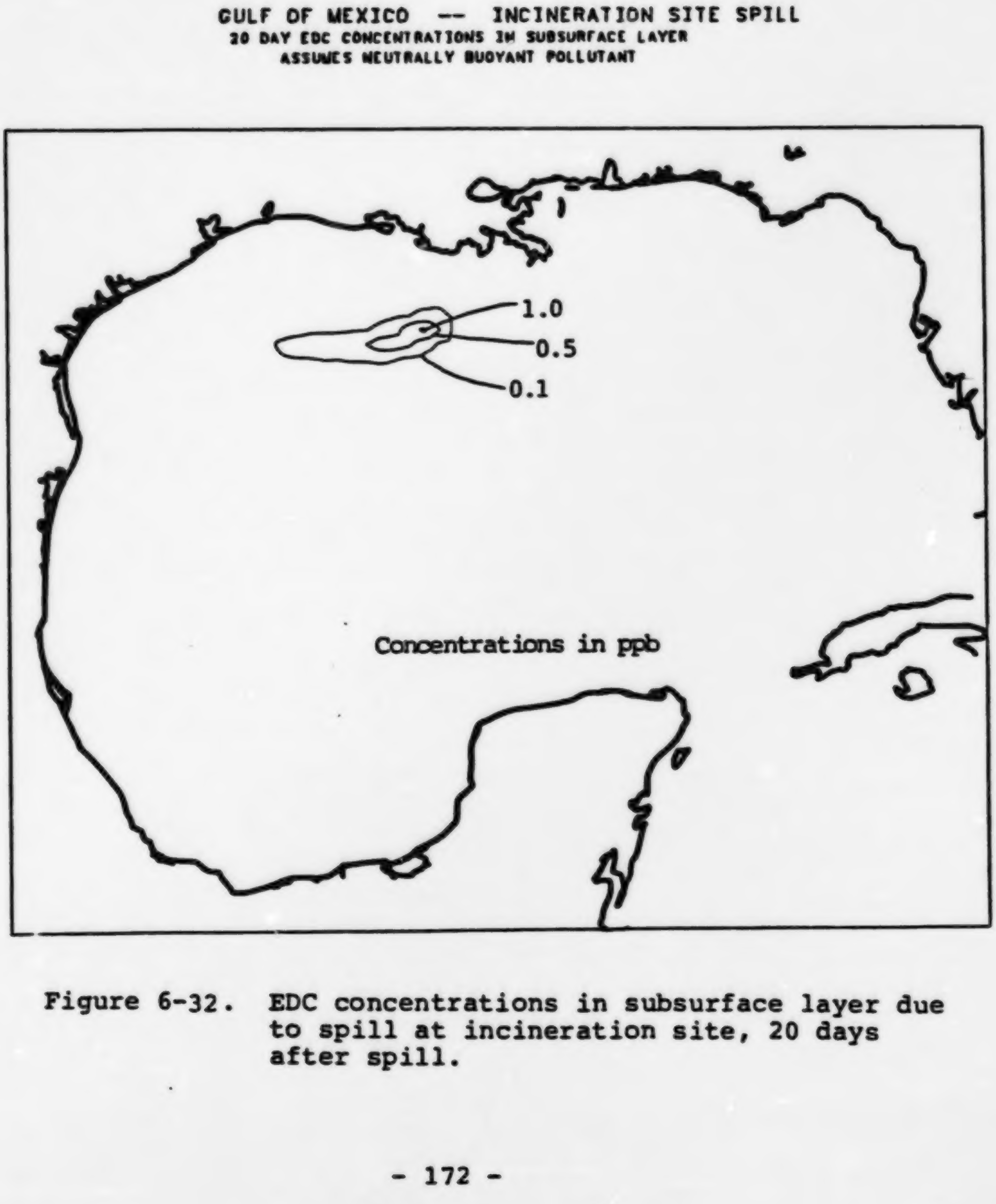

GULF OF MEXICO -- INCINERATION SITE SPILL
20 DAY EDC CONCENTRATIONS IN SUBSURFACE LAYER
ASSUMES NEUTRALLY BUOYANT POLLUTANT
1.0
0.5
0.1
Concentrations in ppb

Figure 6-32. EDC concentrations in subsurface layer due to spill at incineration site, 20 days after spill.

6.3 A Note on Volatilization and Weathering of PCB and EDC

Lyman et al. (1982) gives methods for estimating volatilization rates of dissolved contaminants from water. Based on a Henry's Law constant of 2.7×10^{-3} atm - m^3/mol for Auroclor 1254, as derived from laboratory work (MacKay et al. 1979), the half life T for dissolved PCB in water can be estimated from

$$T = 0.69 \times Z/2$$

where T is in hours and Z is water depth in centimeters. Since this method assumes instantaneous vertical mixing, the validity of the estimate generally decreases with increasing depth, and is probably not valid for sinking spills except in Mobile Bay, where T is about 100 hours. Note that volatilization affects only the dissolved phase of the pollutant, and since the vast majority of PCBs in the water column will appear as individual droplets or will be adsorbed to suspended particulate matter, volatilization will have little effect on total equilibrium concentrations.

A combined laboratory and modeling analysis of evaporation and dissolution (weathering) of actual PCB wastes has been performed by JRB (1984). The work is based on laboratory distillation of several PCB wastes, so that specific volumetric (or mass) fractions of the total waste are distributed along a discretized true boiling point (TBP) scale. The waste is thus modeled as a series of "cuts". Through standard mass transfer equations (see for example Lyman et al. 1982), and empirical correlations for oil spills at sea, weathering can be modeled as a function of wind speed and a prevailing environmental temperature (Payne et al. 1984).

The lighter molecular weight fractions, or cuts, will evaporate and disperse into the water column most rapidly and to the greatest extent, as shown in Figures 6-33 and 6-34. Figure 6-33a shows the mass loss rate due to dispersion for three boiling point cuts of a PCB waste, and for the waste slick as a whole. The lighter cut (415 °F) contains primarily ordinary hydrocarbons. The PCBs are contained in the heavier cuts (554 °F and 712 °F). In comparing the mass fraction dispersed (Figure 6-33a) with that evaporated (Figure 6-33b), it is clear that evaporation dominates the lower boiling point (non-PCB) portions of the waste. As Table 6-3 shows, about one third of the slick (by volume) is associated with these lower boiling point cuts.

Figure 6-34 shows mass balances for a 7.7 m/sec (15 knot) wind. Comparison with Figure 6-33 for a 1.0 m/sec (2 knot) wind shows the strong influence of wind speed on weathering rates. Not only does evaporation increase nearly linearly with wind, but dissolution increases as the wind speed squared (Payne et al. 1984, Spaulding et al. 1982). The conditions of Figure 6-34, 90 °F and 7.7 m/sec, are reasonable for the Gulf of Mexico. From Figure 6-34c, it can be estimated that about 30% to 40% of the PCB-containing portions of a surface slick will be removed within 5 days of release, due primarily to dispersion. Thus the

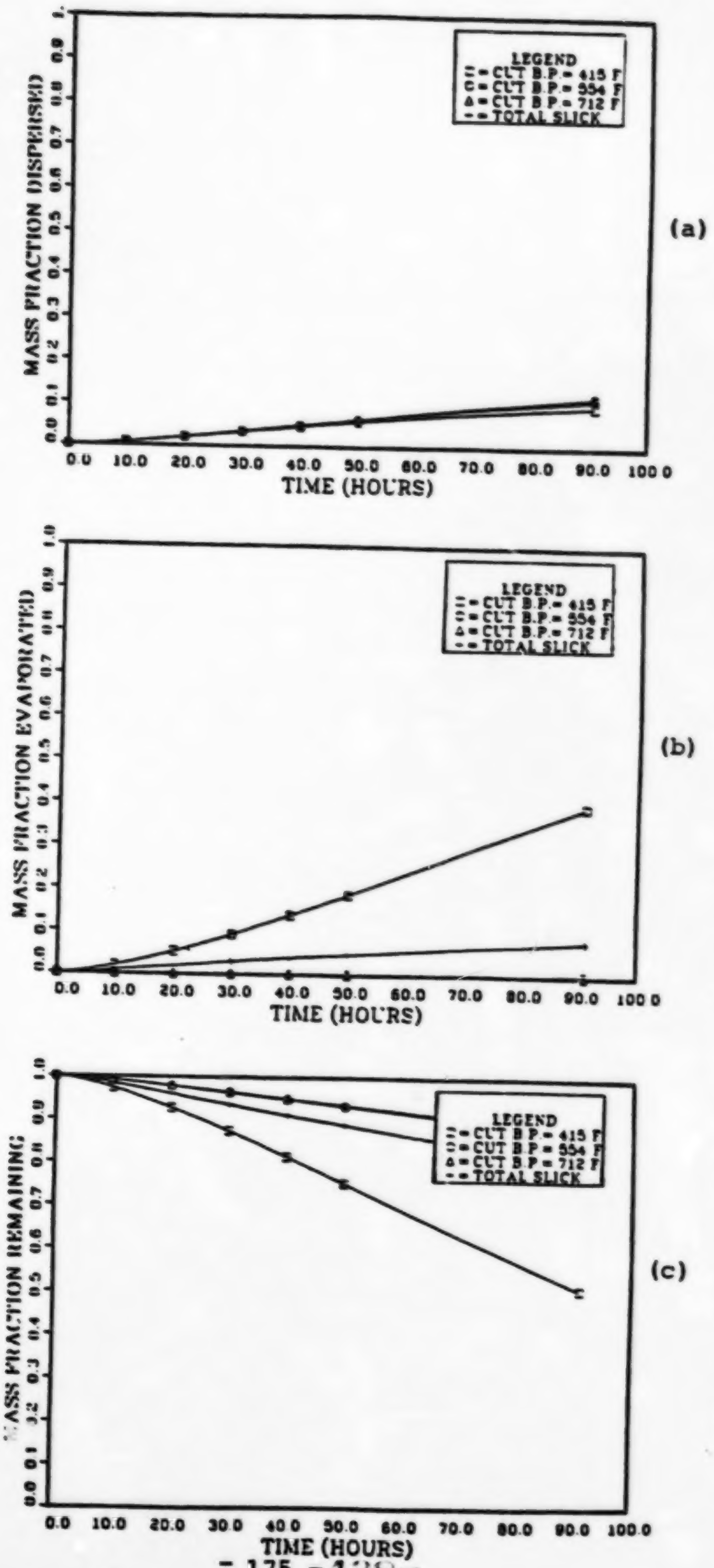


Figure 6-33. Modeled weathering of PCB waste slick at 90° F and a wind speed of 1.0 m/sec (from JRB 1984).

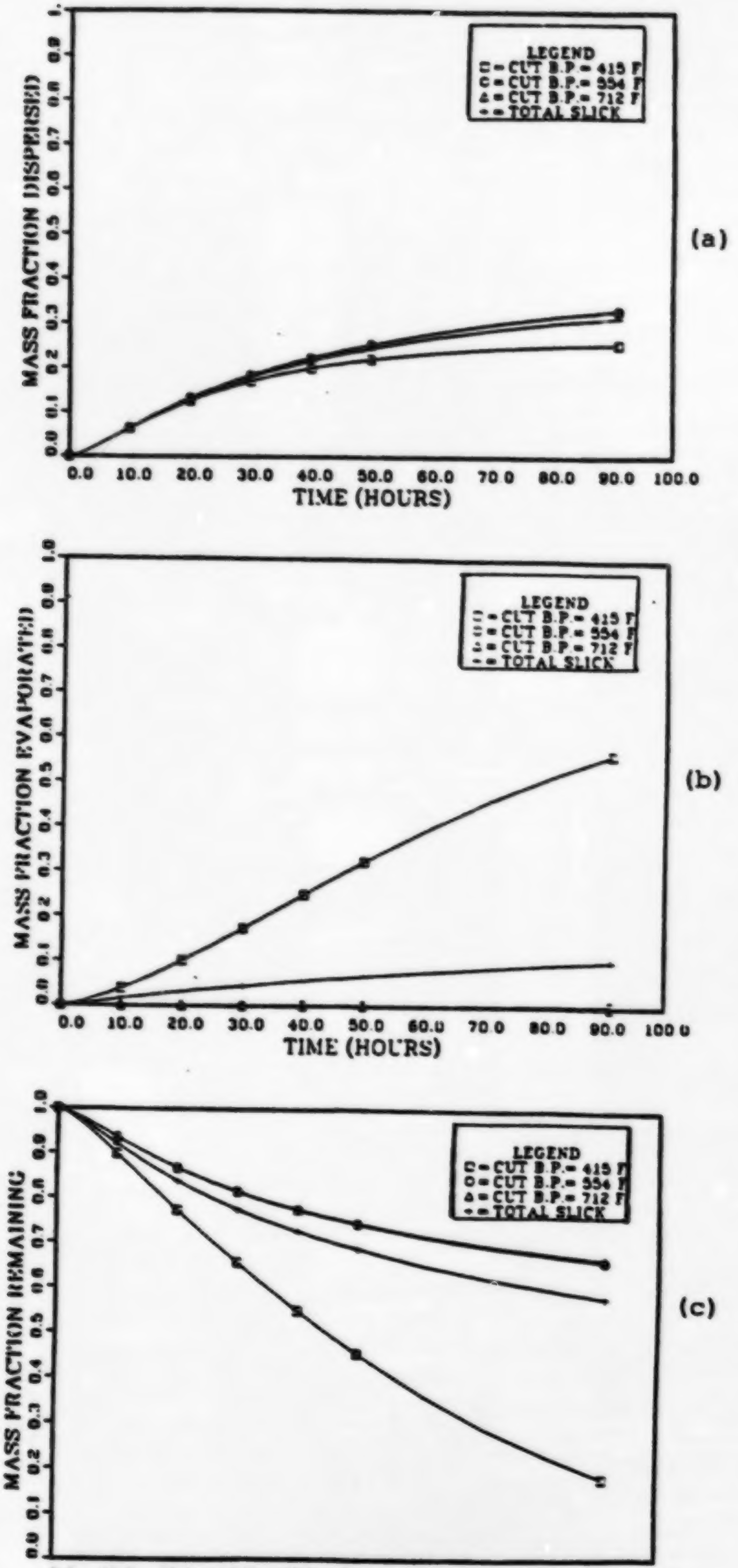


Figure 6-34. Modeled weathering of PCB waste slick at 90° F and a wind speed of 7.7 m/sec. (from JRB 1984)

Table 6-3. Assumed characteristics of a floating PCB waste (from JRB 1984)

Cut #	Boiling Point (°F)	API Gravity	Volume (%)
1	324	22.0	1.77
2	370	21.0	1.77
3	415	20.0	7.38
4	460	19.0	17.6
5	505	18.0	4.68
6	554	17.0	4.57
7	609	16.0	6.55
8	662	15.0	9.88
9	712	14.0	22.8
10	764	13.0	20.8
11	820	12.0	2.20

amounts actually reaching shore from an offshore spill will be considerably smaller than those modeled in Section 6.

For EDC, with a Henry's Law constant of 1.1×10^{-3} atm - m^3/mol, we get a half life estimate for Mobile Bay of about 60 hours. Since EDC is highly soluble, and since the volatilization half life is slightly less than the tidal flushing half life of about 140 hours (Figure 6-28), volatilization is an important consideration. The concentration C in the water can be estimated as:

$$C = C_o \exp (-Kt/Z - K_2t)$$

where C_o is initial concentration, K is the total liquid phase resistance (equal to about 3 cm/hr following estimation procedures given in Lyman et al. 1982), t is time in hours, Z is depth (equal to about 300 cm in Mobile Bay outside the shipping channel), and K_2 is the fractional flushing rate (0.005 per hr.). Comparison of the flushing rate with the volatilization rate K/Z shows that volatilization occurs about twice as rapidly for EDC.

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